Annex to the review of the research and scientific understanding of drought

Chief Scientist's Group report

November 2023

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Dr Robert Bradburne
Chief Scientist

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Introduction

The Environment Agency's Chief Scientist's Group commissioned a series of reviews of existing research into drought in the UK, including the drivers, impacts and management of drought, both now and in the future. The reviews focused on three themes: the physical processes that drive droughts; the impacts of droughts; and the management of droughts. Each theme was further divided into separate topics and each topic was the subject of an individual review undertaken by experts providing answers to questions about drought in the UK, including:

- What is the observed experience of drought?
- What is the current understanding about the causes and variability of droughts?
- What is the current approach to drought management?
- How do we currently model drought and how might droughts change in the future?
- What is known about drought impacts now and expected in the future?

The review group included more than 40 academics from 13 different universities, research institutes, and consultancies. Authors were encouraged to review the existing literature on a specified topic and to comment on what is known and where there are knowledge gaps and uncertainties. To gain the widest range of ideas, authors were given freedom to form their own views without editorial control from the Environment Agency.

An overview of the reviews is presented in Environment Agency (2023). Review of the research and scientific understanding of drought. Environment Agency, Bristol. This report is an annex to the summary report and presents the essays and overviews as written by the authors, with only minor editing of the format.

A: Atmospheric Circulation, Low Rainfall, and Drought in the UK

Len Shaffrey, National Centre for Atmospheric Science, University of Reading.

Overview

Droughts have severe impacts on societies, economies, agriculture and ecosystems. Recent events in the UK, including the 2010-2012 drought and drought conditions in 2018 and 2022, have highlighted the need to revaluate our understanding of droughts. From a meteorological perspective, the maritime continent of the UK is characterised by the passage of rain-bearing storms embedded in the westerlies over the North Atlantic Ocean. Prolonged periods of low rainfall are generally associated with fewer storms passing over the UK, often due to anticyclonic blocking events. Furthering our understanding of how droughts have varied in the past and may change in the future requires a better understanding of how the variability in atmospheric circulation leads to low rainfall, and how it might respond to climate change. This includes addressing questions such as:

How has the occurrence of low rainfall and drought in the UK varied in the historical record? Historical records enable a view of UK drought over two and a half centuries. During that time, the UK has experienced lengthy periods of low rainfall, including the 'long droughts' in the 19th Century (1854-1860, and 1890-1910). We currently have a poor understanding of the atmospheric drivers of the long droughts, which raises questions whether we fully understand the risk of such events were they to occur today. In addition, the extent to which recent trends in North Atlantic atmospheric circulation, and associated impacts on UK rainfall, are due to external forcing (greenhouse gases, anthropogenic aerosol, etc..) versus internal variability are still not clear. Addressing these issues will require investment in recovering old weather records and developing long-term atmospheric reanalysis. In addition, the reanalysis of the past needs to be combined with multiple lines of evidence (meteorological, hydrological, impacts) to provide the best assessment of historical drought.

What are the prospects for improving early warnings of drought on s2d (subseasonal-to-decadal) timescales? There have been recent improvements in s2d dynamical forecasts of the atmospheric circulation, for example, skilful seasonal forecasts of the wintertime North Atlantic Oscillation are now possible. However, there remain substantial challenges and barriers to the uptake of s2d forecasts by water resource managers. A key question is whether further improvements could be obtained by coupling s2d forecasts of the North Atlantic atmospheric circulation with hydrological forecasts and new machine learning approaches.

What impact could climate change have on the occurrence of low rainfall and drought in the UK, and associated variability in atmospheric circulation? Climate change is expected to make UK winters warmer and wetter and summers hotter and drier. The increase in rainfall in winter is partly associated with warmer temperatures and thus more moisture and rainfall, and partly associated with an extension of the North Atlantic jet stream over the UK and thus more storms. In summer, the decrease in rainfall is associated with the northwards shift in the jet stream, which leads to a reduction in rainfall. In addition, hotter summer temperatures could be associated with an increase in evaporation. However, there is substantial uncertainty in climate model projections, which in part may be due to climate model biases in atmospheric circulation, such as an underestimation of the frequency of blocking patterns. Thus there is a need to improve climate models and the representation of key processes. For example, there is evidence that higher resolution climate models may help reduce atmospheric circulation biases. Despite these biases, large ensembles of climate model simulations might be helpful in generating event sets of modelled droughts to help inform current and future drought risk assessments, especially the 1-in-500 year events required for planning in water resource management. In addition, approaches such as Storylines could be employed to help gain insight into future UK drought risk.

1. Introduction

1.1. Motivation

Droughts have severe impacts on societies, economies, agriculture and ecosystems. For example, the 1975-76 drought had a devastating effect on the UK economy causing an estimated £3,500m loss to agriculture, £700m of subsidence damage to buildings and a £400m cost to the water industry (figures adjusted for inflation, Rodda and Marsh, 2011). Recent events in the UK, including the 2010-2012 drought and drought conditions in 2018 and 2022, have highlighted the need to revaluate our understanding of droughts. It is essential to understand the current risks of UK drought and how those risks might change in the future due to climate change.

This essay will focus on UK drought from the perspective of precipitation in the UK, the North Atlantic and European atmospheric circulation, and larger-scale drivers. From a meteorological perspective, the maritime continent of the UK is characterised by the passage of rain-bearing storms embedded in the westerlies over the North Atlantic ocean. Prolonged periods of low rainfall are generally associated with fewer storms passing over the UK, often due to anticyclonic blocking events. Furthering our understanding of how droughts have varied in the past and may change in the future therefore requires a better understanding of how the variability in atmospheric circulation drives low rainfall, and how it might respond to climate change.

1.2. Aim and Research Questions

The aim of this essay is to discuss the latest understanding of atmospheric circulation and its variability in the context of low rainfall and drought in the UK. The research questions considered are:

- How has the occurrence of low rainfall and drought in the UK varied in the historical record? And how do variations in atmospheric circulation affect the occurrence of low rainfall and drought?
- What are the prospects for improving early warnings of low rainfall and drought on s2d (subseasonal-to-decadal) timescales?
- What impact could climate change have on the occurrence of low rainfall and drought in the UK, and associated variability in atmospheric circulation?

1.3. Outline

Given the research questions outlined above, the essay is structured as follows:

Section 2. Atmospheric variability in the context of low rainfall and drought in the UK: This section will outline current understanding of the variability in the atmospheric circulation that results in low rainfall and drought in the UK and Europe.

Section 3. Historical low rainfall and drought in the UK: Section 3 will discuss the key observational evidence for low rainfall and drought in observational records.

Section 4. Subseasonal-to-decadal variability and predictability of low rainfall and drought in the UK: Section 4 will summarise the current understanding of the drivers of s2d (subseasonal-to-decadal) variability in atmospheric circulation in the North Atlantic and Europe. Section 4 will also discuss prospects for developing skilful s2d forecasts.

Section 5. Climate change, atmospheric circulation and low rainfall and drought in the UK: Section 5 overviews the latest understanding of the impacts of climate change on low rainfall and atmospheric circulation in the context of UK drought. Given the difference in seasonal responses, the impacts of climate change will be considered separately for winter and summer.

Section 6. Summary and research questions.

Note that as the discussion covers a lot of ground and is rather wide ranging, this essay is not a comprehensive review article. Therefore only selected references are cited. Note also that this essay will focus on UK low rainfall and links to atmospheric circulation, which is related to the similar concepts of meteorological drought and precipitation deficits.

2. Atmospheric variability in the context of low rainfall and drought in the UK

In section 2, the atmospheric variability over the UK and Europe in the context of low rainfall and drought will be discussed. Section 2.1 will describe some of the common ways of characterising the atmospheric circulation, while section 2.2 discusses long-term trends seen in the observations. The maritime climate of the UK is characterised by the passage of extratropical storms embedded in the westerly atmospheric flow over the North Atlantic Ocean. The precipitation associated with extratropical storms and frontal systems accounts for over 90% of the wintertime precipitation over the UK (e.g. Hawcroft et al. 2012). Consequently, prolonged periods of low rainfall are generally associated with fewer storms passing over the UK. However, the variability in regional rainfall within the UK, and its association with the large-scale atmospheric circulation, is more complex (e.g. Wilby et al., 1997). This will be addressed further in section 2.3.

2.1. Characterising the North Atlantic and European Atmospheric Circulation

Section 2.1 will outline some of the methods used to characterise the atmospheric circulation over the North Atlantic and Europe. Broadly, these methods can be separated into approaches focusing on physical phenomena (blocking, jet streams, etc.) and approaches which use statistical methods.

2.1.1. Phenomenological approaches to characterising the atmospheric circulation

Here the characterisation of the atmospheric flow is based upon physical atmospheric phenomena most relevant during observed periods of low rainfall, for example the occurrence of anticyclonic blocking over the UK and Northwest Europe, and the position and strength of the North Atlantic jet stream.

Anticyclonic Blocking: Low rainfall over the UK is often associated with anticyclonic conditions (that is high surface pressure), where the prevailing south-westerly flow is substantially weakened or even reversed (Woollings et al. 2018). Blocking preferentially occurs in a few regions over the Northern Hemisphere, including over Europe at the end of the North Atlantic storm track. The seasonal cycle of blocking frequency is relatively small, and persistent blocking events over Europe can occur in all seasons (Schiemann et al. 2020).

Although blocking over the UK in winter and summer is associated with low rainfall, the seasonal impacts can be different. In winter, blocking over the UK is associated with the advection of cold air from continental Europe and cold temperatures. In summer, blocking is associated with the advection of hot air from the continent, clear skies, enhanced solar radiation and therefore hot temperatures. In summer, this can lead to enhanced evaporation, drier soils, and increased water demand, all of which can exacerbate drought conditions (Kautz et al. 2022).

There is no standard definition of blocking, however, common characteristics of blocking include “persistence, quasi-stationarity and obstruction of the usual westerly flow and/or storm tracks” (Woollings et al., 2018). Various indices are used to characterise blocking, which generally identify either long-lived anomalies or reversals in meridional gradients of mid-to-upper level tropospheric meteorological variables. These include 500-hPa geopotential height (Tibaldi and Molteni 1990), the vertically averaged potential vorticity (Schwierz et al. 2004), or the potential temperature on the PV=2 (potential vorticity=2) surface (Pelly and Hoskins, 2003). The various flavours of blocking indices means that it can be difficult to identify consistent statistics between studies (Barnes et al. 2012, 2014; Woollings et al. 2018).

North Atlantic Jet Stream Strength and Latitude: Various indices have been designed to measure the mean latitude and strength of the jet stream, for example, the mean strength and latitude of the lower tropospheric, westerly winds over the North Atlantic (60oW to 0oW, Woollings et al. 2010). During prolonged periods of low rainfall in the UK, the jet stream is often deflected northwards or southwards from its climatological position, steering storms and their associated rainfall away the UK. Large northward or southward deflections of the North Atlantic jet stream are often associated with blocking and Rossby wave breaking (Masato et al. 2012).

2.1.2. Statistical approaches to characterising the atmospheric circulation

An alternative way of characterising the atmospheric circulation is to use statistical methods. Section 2.1.2 describes the statistical methods for charactering the atmospheric circulation over the North Atlantic and Europe that are most frequently employed in the literature. To ensure that results are not an artifact of the statistical method, it is essential results are interpreted and assessed to ensure they are physically realistic (Ambaum et al. 2002).

Correlation and Regression maps: One of the simplest ways to ascertain the characteristics of the atmospheric circulation associated with low rainfall in the UK is to correlate or regress maps of meteorological variables against an index of a variable of interest. It also is possible to determine the preferred patterns of variability in atmosphere circulation by correlating or regressing meteorological variables, such as MSLP (mean sea level pressure) or 500hPa geopotential height, with their variation at a single point (Wallace and Gurtzler, 1981, Raible et al. 2014). These *one-point correlation or regression maps* when applied to the North Atlantic and European region reveal that variations in wintertime MSLP to the south of the North Atlantic jet stream are anticorrelated with variations to the north of the jet stream. The correlated variations are known as the North Atlantic Oscillation (NAO), which has centres of action in the subtropics nears Azores and in the subpolar regions near Iceland (Hurrell, 1995, Jones et al., 1997b). An NAO index can be defined as the normalised pressure difference between the Azores and Iceland. When the NAO index is positive, the westerlies over the wintertime North Atlantic are strengthened, which bring storms to the UK and results in increased mean UK rainfall. When the NAO index is negative, the westerlies are weakened, resulting in fewer storms and decreased mean UK rainfall.

Empirical Orthogonal Functions and Singular Value Decomposition methods: Another method to characterise the atmospheric circulation over the North Atlantic and European region is to use Empirical Orthogonal Functions (EOFs) and similar Singular Value Decomposition (SVD) methods. An EOF analysis of the winter North Atlantic MSLP decomposes the MSLP into eigenvectors which are orthogonal in space and time. The eigenvector that explains the largest variance is referred to as the First EOF, the second largest variance the Second EOF etc. The First EOF pattern in the winter MSLP over the North Atlantic and Europe very closely resembles the North Atlantic Oscillation pattern found in correlation analyses, and generally explains 35%-50% of the winter MSLP variance (e.g. Figure 2.1, from Comas-Bru and McDermott, 2014).

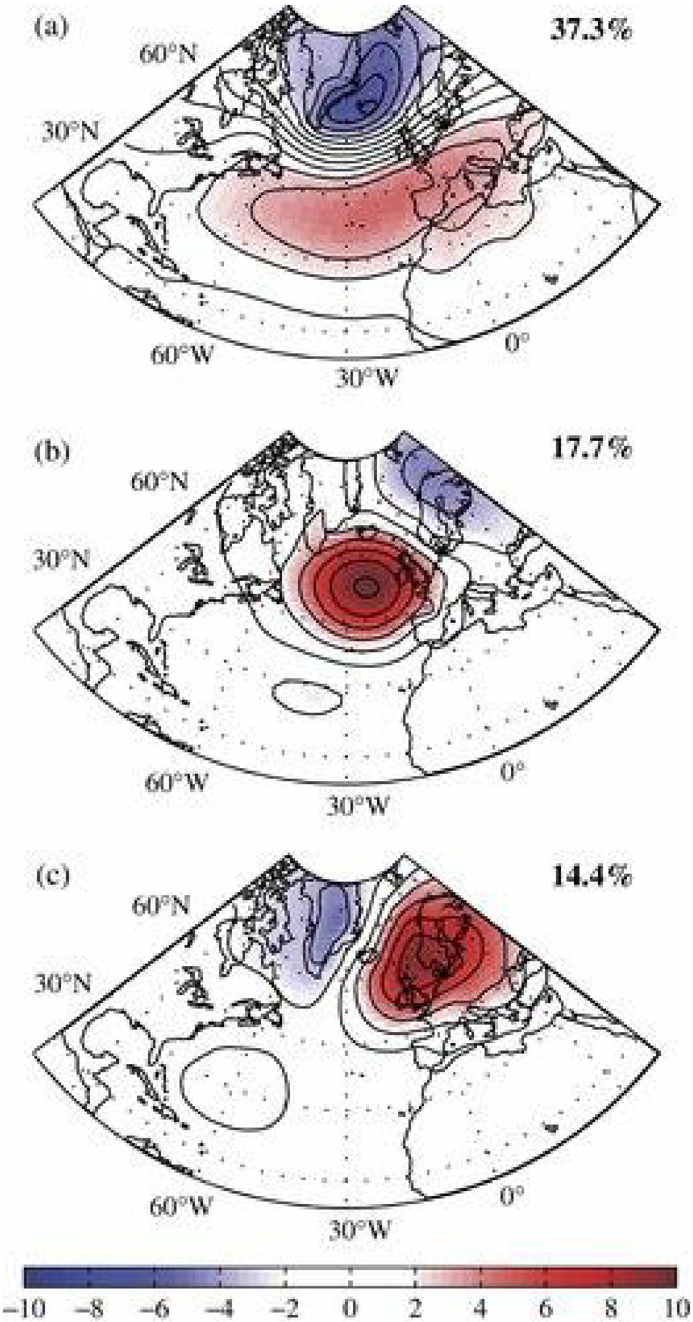

Figure 2.1: Spatial maps of the first three eigenvectors of the gridded winter (December–February) monthly sea-level pressure anomalies (in mb) for the North Atlantic domain (1872–2009) calculated using the 20CRv2 global dataset (Compo et al., 2011): (a) North Atlantic Oscillation (NAO), (b) East Atlantic pattern (EA) and (c) Scandinavian pattern (SCA). From Comas-Bru and McDermott (2014).

Figure 2.1 also shows the Second and Third EOFs of winter MSLP, which typically explain 15-20% and 10-15% of the variance respectively. The Second EOF is often named the East Atlantic Pattern, and is closely related with higher surface pressures and blocking patterns to west of the UK. The Third EOF is named the Scandinavian pattern, and is associated with higher surface pressures and blocking patterns north east of the UK. In summer, the leading EOF patterns are similar to those in winter, but tend to explain less variance and have slightly different spatial patterns. The SNAO (Summer North Atlantic Oscillation, Folland et al. 2009) is similar to the winter NAO but is displaced northwards relative to winter (associated with the northward seasonal migration of the North Atlantic jet stream in summer).

North Atlantic Weather Regimes and Weather Types*:* Previous studies to have attempted to determine preferred patterns of variability in atmospheric circulation using methods that identify similar synoptic weather regimes or types of circulation. These methods tend to be more complex than the approaches described above, and produce a larger number of patterns of variability, which are often referred to as weather regimes or weather types. Broadly, these methods can be split into two categories. Firstly, are methods that identify a few key weather regimes using objectives techniques such as k-means clustering of EOF patterns (e.g. Corti et al. 1999, Dawson et al. 2012).

Secondly, are methods that identify a larger range of patterns or weather types either by expert judgement or by more complex statistical techniques. Methods to characterise the atmospheric circulation by expertly examining the day-to-day variations of synoptic weather maps have long been established for the UK (e.g. Lamb's Weather Types; Lamb, 1950) and central Europe (the Grosswetterlage developed during the 1940s and 1950s by Baur, Hess and Brezowsky). Objective weather typing methods have seen a recent resurgence due to the development of new statistical clustering techniques, for example the Met Office Weather Types used in the UKCP18 projections (Maisey et al. 2018), the use of objective Lamb's Weather Types to investigate the relationship between hazards and atmospheric circulation over the UK and Ireland (De Luca et al., 2019), and objective Grosswetterlage to describe the recent 2018 heatwave in central Europe (Hoy et al. 2020). Weather typing analyses tend to produce a large number of weather types, and so a degree of expert judgement is required in order to associate particular weather types with prolonged periods of low rainfall that are associated with drought but can provide useful additional insight (e.g. Pope et al. 2022).

It is worth noting that these very different methods for characterising the atmospheric circulation are often complementary and provide similar views on atmospheric variability and change (e.g. Woollings et al. 2012). Sometimes care and thought needs to be given to reconciling these different perspectives, e.g. information about the variability of the atmospheric circulation on different timescales where eddy-mean feedbacks are important (such as daily and seasonal timescales).

2.2. Trends in Atmospheric Circulation over the North Atlantic and Europe

The extent to which trends exist in the atmospheric circulation over the North Atlantic and Europe is an area that has received substantial study, especially in the context of assessing the impacts of climate change. However, the seasonal-to-multi-decadal variability in the North Atlantic and Europe circulation is large, which generally makes it very difficult to detect long-term trends due to the relative shortness of robust observational records.

Over the past few decades, there has been an increasing investment in data recovery of observational records, especially in terms of recovering and digitising old weather records (e.g. the ACRE project, Allan et al. 2011). Efforts in data recovery help to inform past assessments of previous droughts, especially when combined with additional lines of observational evidence (e.g. past hydrological observations and textual analysis of written records, e.g. Dayrell et al. 2022). In addition, long-term atmospheric reanalyses, such as the 20th Century Reanalysis (Compo et al. 2011, Slivinski et al. 2019) and the ECMWF ERA20C and CERA20C reanalyses (Poli et al. 2016), have recently been developed, which provide additional information about the historical evolution of the atmospheric circulation. Long-term reanalysis primarily assimilates surface observations (primarily sea surface temperatures, surface pressures, and in the case of ERA20C and CERA20C, marine surface winds) and statistically combine them with numerical weather prediction models to determine the atmosphere state. However, the assimilation of only surface observations can result in substantial uncertainties in estimating the state of the atmospheric circulation.

Given the large variability in the mid-latitudes, long-terms trends in the atmospheric circulation are difficult to detect. This is indicated in Figure 2.2, which shows the Winter NAO and Summer NAO from Kendon et al. (2022) for 1850 to present, which shows large interannual and decadal variability but no long-term trends. However, there are previous studies which have posited there are long-term trends in other aspects of the atmospheric circulation. For example, Donat et al. (2011) found an upward trend in storminess over Northern Europe in the 20th Century Reanalysis (v2) from 1879 to 2010 (Compo et al. 2011). This trend was further investigated in Kruger et al. (2013, 2019), which found there was no trend in the direct MSLP observations and this was inconsistent with the increased storminess seen in the 20th Century reanalysis (v2). Varino et al. (2018) found significant upward trends in 20th Century storminess over the North Atlantic and Europe in the ERA20C reanalysis. Closer analysis between observed surface winds and those in ERA20C showed that potentially spurious trends in storminess were being generated through the assimilation scheme (Bloomfield et al. 2018, Wohland et al. 2019). Cornes et al. (2013) also investigated circulation indices using MSLP observations, including a Paris-London westerly index back to 1692. Cornes et al. (2013) found little evidence of trends, although noted that summer westerlies from the 1970s had become unusually weaker. This may be associated with the low values of the SNAO seen over the past few decades (Kendon et al. 2022, Figure 2.2).

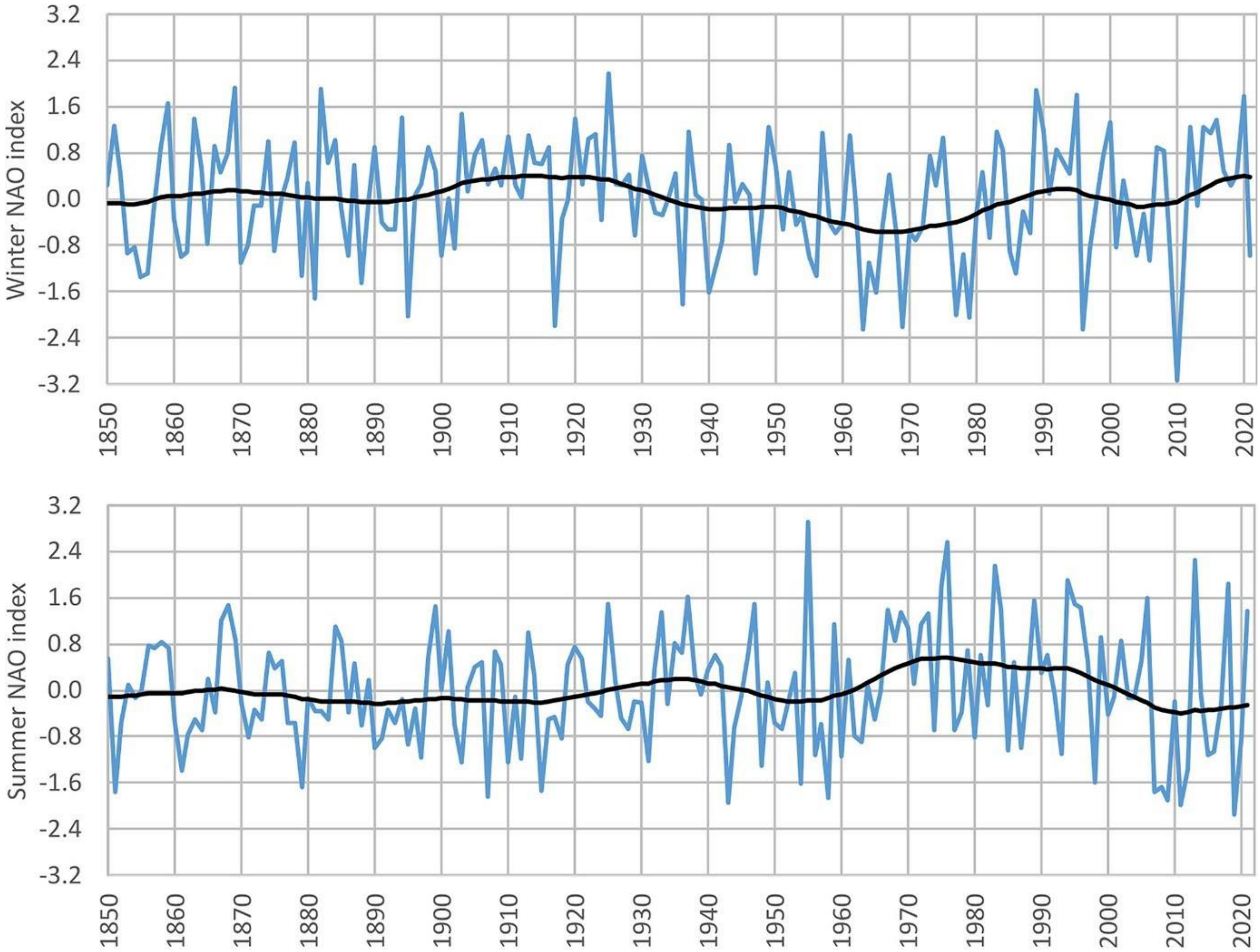

Figure 2.2 Upper) Winter NAO index based on standardized monthly mean pressure difference between stations in Gibraltar and southwest Iceland. Winter 2021 refers to the period December 2020–February 2021. Lower) Summer NAO index based on standardized monthly mean pressure difference using the 20th century reanalysis (Slivinski et al., 2019) and extended to the present day using the ERA5 reanalysis. From Kendon et al. (2022).

For anticyclonic blocking, Barnes et al. (2014) found no trends in North Hemisphere blocking over the past few decades in a variety of modern atmospheric reanalyses for a number of different measures. However, Hanna et al. (2022) reported a very recent increase in blocking around Greenland in summertime using MSLP observations. Although Greenland blocking may not directly influence the circulation over the UK may be indicative of wider changes in the North Atlantic circulation. Additionally, Wazneh et al. (2021) reported finding North Atlantic blocking becoming more persistent in summer and autumn in the CERA20C reanalysis, but these results will need to be verified using MSLP observation given the previously reported issues with CERA20C (Wohland et al. 2019). The equinoctial seasons are often less studied than summer and winter in this context, but Cotterill et al. (2021) used a weather typing analysis and found evidence of a positive trend from the 1970s in autumn anticyclonic weather types, associated with drier conditions. In summary, there is not much evidence of long-term trends in atmospheric circulation, and recent work has highlighted the need to reappraise long-term datasets and reanalyses. There is, however, some evidence that recent variability in the summer North Atlantic circulation might be considered unusual from a historical perspective.

2.3. Relationship between Regional UK Rainfall and Atmospheric Circulation

As mentioned in the introduction to section 2.1, the relationship between the regional rainfall in the UK and atmospheric circulation is more nuanced than that for between the mean UK rainfall and atmospheric circulation. For example, Wilby et al. (1997) showed that for winters with a strong positive NAO index, the west of Scotland had the strongest positive rainfall anomalies, while eastern England had negative rainfall anomalies. In contrast, in years with a strong negative NAO index, eastern England had positive rainfall anomalies while the west of Scotland had negative rainfall anomalies. This suggests that different patterns of variability in atmospheric circulation govern variation in regional rainfall in the UK.

This was also seen in Lavers et al. (2010), who found that winter precipitation in the north-western UK was correlated with westerly winds and a MSLP dipole similar to the NAO. For the south-east of England, winter precipitation is correlated with negative MSLP anomalies centred over the UK and westerly winds to the south, a pattern in strong agreement with the East Atlantic pattern. Similar results for wintertime were found in Hall and Hannah (2018) and Baker et al. (2018). In addition, Baker et al. (2018) found that for summer that regional rainfall in the UK is primarily associated with the latitude of the westerlies over the North Atlantic.

3. Historical periods of low rainfall and drought in the UK

Our understanding of past variability of low rainfall periods and drought is contingent on the historical records of precipitation from weather records. Section 3.1 discuss the historical records of precipitation in the UK in the context of low rainfall. Section 3.2 discusses different methods to characterise low rainfall and drought.

3.1. Observational records of rainfall in the UK

There has been substantial investment in the recovery, analysis, and quality control of rain gauge data in the UK. For example, the timeseries from 1836 of seasonal rainfall from the HadUK-Grid dataset is shown in Figure 3.1 (Hollis et al. 2019, Kendon et al. 2022). Values are expressed as percentage anomalies from the 1991-2020 average. There is large interannual variability in all four seasonal timeseries, but also an indication of decadal variability. There has been a marked increase in winter rainfall across the timeseries in the most recent decade with 2014, 2016 and 2020 all in the top five wettest (the other winters being 1995 and 1990). In summer, there is no indication of long-term changes in the time series. However, it is worth noting that the most recent decade was wetter than the relatively dry summer period from the 1960s until 2000. This may be associated with the changes in the SNAO discussed in section 2.

Longer timeseries of rainfall records are available such as the EWP (England Wales Precipitation) timeseries. EWP is notable for having continuous monthly records from 1766, and forms part of the HadUKP dataset (Alexander and Jones, 2001). Precipitation in EWP shows trends towards wetter winters and drier summers (Jones and Conway, 1997a). However, Murphy et al. (2020a) found that winter records before 1870 are likely biased low due to the under-catch of snowfall. Murphy et al. (2020a) also found that summer records before 1820 were biased high, which was attributed to decreasing network density and less certain data at key stations. Furthermore, a significant trend to drier summers is not found when EWP series was reconstructed using independent predictors. These results again highlight the ongoing need to reappraise long-term datasets.

With these caveats in mind, precipitation records can be used to identify periods of low rainfall in the UK. For example, Marsh et al. (2007) identified prolonged periods of low rainfall and droughts in the UK from 1800-2006. More recent work has reassessed the 'forgotten drought' of 1765-1768 (Murphy et al 2020b). Rainfall records, especially when combined with hydrological records (Barker et al. 2016, 2019) and other records of impacts (Dayrell et al. 2022) can provide additional insight into the past variability of rainfall and drought in the UK. A historical view also underlines that the nature of droughts in the UK can differ from those experienced in the past few decades. For example, the long droughts of 1854-1860 and 1890-1910 would present different challenges to the resilience of present-day water resource infrastructure. However, there are major knowledge gaps regarding the likelihood that similar long drought events might reoccur, and their relationship to the decadal variability in North Atlantic and European climate and atmospheric circulation.

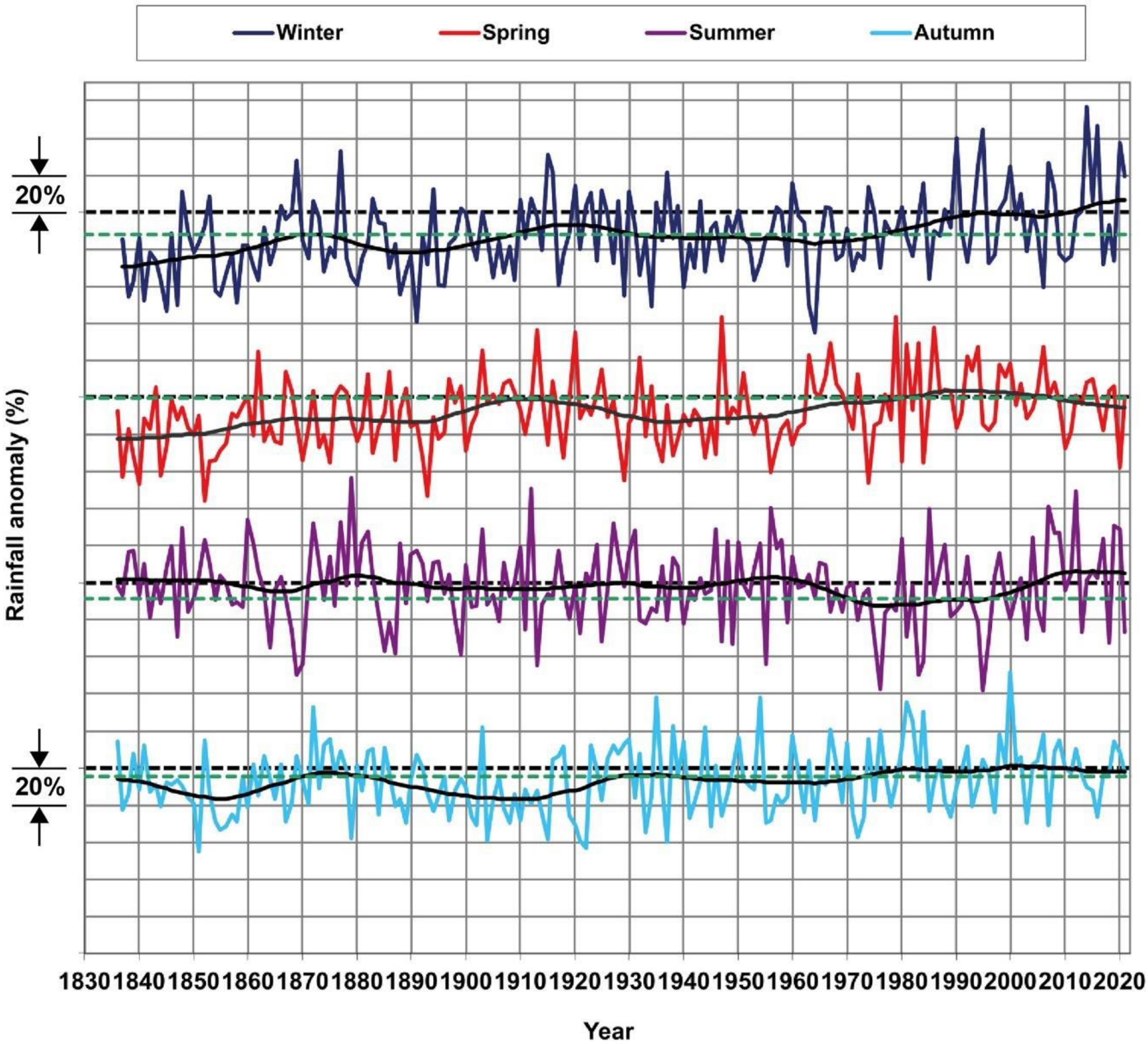

Figure 3.1 Seasonal rainfall for the UK, 1836–2021 (note winter from 1837 to 2021; year is that in which January and February fall.) from HadUK-Grid. The hatched black line is the 1991–2020 long-term average. The lower hatched green line is the 1961–1990 long-term average. Light grey grid-lines represent anomalies of ±20%. The table provides average seasonal rainfall values (mm). From Kendon et al. (2022).

3.2. Characterising low rainfall and drought

Many measures have been developed to characterise low rainfall and drought using rainfall (WMO, 2016) and other records, such as hydrological observations (e.g. see the EA report by Hannaford et al., 2023). Droughts are often measured as deviations from climatological norms, and norms can be difficult to obtain, especially in a changing climate. This has led to a wide range of drought measures being developed and raised questions about whether defining a universal drought indicator is impractical (Lloyd-Hughes, 2014). Nonetheless, section 3.2 will briefly outline some measures to define low rainfall and drought. The Standardized Precipitation Index (SPI, McKee at al. 1993) is recommended by the WMO as a starting point for characterising low rainfall and meteorological drought. SPI can be interpreted as the number of standard deviations by which the observed anomaly deviates from the long-term mean and can be created for differing periods (typically 1-to-36 months) using monthly input data. There are many other indices which characterise different aspects of low rainfall and drought. For example, CDD (Consecutive Dry Days, Karl et al 1999) defined as the number of consecutive days with rainfall less than 1mm day^{-1}.

Although the focus in this essay is on atmospheric circulation and low rainfall, many drought indices incorporate temperature and evaporation. An early operational drought index is the Palmer Drought Severity Index (PDSI, Palmer, 1965, Alley 1984). Another example is the Standardized Precipitation Evapotranspiration Index (SPEI, Vicente-Serrano et al. 2010), which uses the difference between precipitation and potential evapotranspiration to represent a simple climatic water balance. Similar to SPI, SPEI can be calculated for differing periods. It is worth noting that trends in SPI and SPEI differ in observations in Europe (Stagge et al. 2017). These differences are primarily driven by increases in temperature that change the reference evapotranspiration in the SPEI index.

4. Subseasonal-to-decadal variability and predictability of low rainfall and drought in the UK

Over the past decade there has been substantial amount of research into s2d (subseasonal-to-decadal) forecasts of midlatitude atmospheric circulation, temperatures and rainfall. Being able to skilfully forecast these variables on s2d timescales would have a substantial impact on the ability to manage and plan for climate risks, e.g. water resource management and drought in the UK. In the tropics, skilful seasonal forecasts of the coupled climate have been available for a number of decades, especially for strongly coupled ocean-atmosphere processes such as ENSO (the El Nino Southern Oscillation). Skilful seasonal forecast in the more variable midlatitudes have been more difficult to produce. However, earlier statistical analysis identified processes that might give rise to seasonal forecast skill (e.g. Colman et al. 1999). More recently, skilful dynamical forecasts have recently been developed in the midlatitudes, e.g. of dynamical seasonal forecasts of the wintertime North Atlantic Oscillation (Scaife et al. 2014). The development of skilful seasonal forecasts may provide new insights into the management of water resources.

Section 4.1 will provide a short overview of the large-scale climate drivers that give rise to s2d variability and predictability in the North Atlantic and European sector. Following this, section 4.2 will discuss the potential for developing skilful s2d forecasts of the North Atlantic and Europe atmospheric circulation and climate system.

4.1. Subseasonal-to-decadal large-scale climate drivers of North Atlantic and European atmospheric circulation in the context of low rainfall and drought

There is a broad body of literature investigating the large scale drivers of the atmospheric circulation and climate variability in the North Atlantic and Europe. In section 4.1, a short overview of the key large-scale drivers is outlined, with a particular focus on low rainfall and drought in the UK. The key large-scale drivers differ between summer and winter, and so will be discussed separately.

Winter: Atmospheric interactions between the extratropics and the tropics play a key role in driving variability in the North Atlantic and European atmospheric circulation. The leading pattern of variability in the tropics is ENSO (the El Nino Southern Oscillation). During the positive phase of ENSO (El Nino), the ocean warms and easterly winds weaken over the Tropical Pacific ocean. This results in atmospheric convection and precipitation over the Maritime Continent moving out into the central Tropical Pacific. Changes in ENSO drive variability in the mid-latitudes i) through the propagation of Rossby waves (Hoskins and Karoly, 1981, Scaife et al. 2017) and ii) indirectly by influencing variability in the Arctic stratosphere (Ineson et al. 2009). Tropical forcing from the Madden Julian Oscillation (Cassou et al. 2008) and from the Indian Ocean (Hardiman et al. 2020) may also have an impact on the atmospheric circulation of the North Atlantic and Europe. Investigations between tropical teleconnections and lowland droughts in the UK also identified extratropical-tropical interactions as a driver of drought in the UK, more specially a statistical link was found between ENSO (during the negative phase known as La Nina) and UK precipitation deficits (Folland et al. 2015). The impact of ENSO has also been identified in Lamb Weather Types over the UK (Wilby, 1993). There is also evidence that the lagged influence of wintertime ENSO signals might influence the springtime rainfall over the North-western Europe (Lloyd-Hughes and Saunders, 2002).

Variability in the stratosphere may also impact on the North Atlantic and European atmospheric circulation. Strong winds (known as the polar vortex) form as the polar stratosphere cools during winter. The polar stratospheric vortex can breakdown and warm very rapidly (a process known as Sudden Stratospheric Warming). Sudden Stratospheric Warmings can induce a weakening and/or deflection of the North Atlantic Jetstream leading to cold and dry anticyclonic blocking conditions over the UK and Europe (Baldwin and Dunkerton, 2001). Sudden Stratospheric Warming events occurred during the winter of 2010 (Dornbrack et al. 2012) and spring 2018 (Roa et al. 2018) as precursors of dry conditions in the UK. Additional aspects of stratospheric variability can also influence the evolution of the atmospheric circulation over the North Atlantic through similar mechanisms, for the example the Quasi-Biennial Oscillation (Andrews et al. 2019).

There has also been a substantial amount of research into the impact of the rapid loss of Arctic Sea ice and warming in the Arctic on the winter mid-latitude circulation. A number of early observational studies suggested a relationship between the reduction of Arctic Sea ice (especially in the Kara-Barents Sea) and an increase in anticyclonic blocking and cold, dry conditions in Europe (Honda et al. 2009). Other studies suggested that warming in the

Arctic might result in a weaker and more “meandering” North Atlantic jet stream that might also lead to an increase in anticyclonic blocking (Francis and Vavrus, 2012). Additional observational studies have not always replicated these finding from early studies (Barnes et al. 2013), and modelling studies have found a robust but weak response (Smith et al. 2022). Recent review articles have highlighted these discrepancies (e.g. Barnes and Screen, 2015, Cohen et al. 2020). Cohen et al. (2020) stated “Divergent conclusions between model and observational studies…continue to obfuscate a clear understanding of how AA is influencing midlatitude weather.”

Variations in the circulation of the North Atlantic Ocean are also thought to have an influence on the climate of Europe and the UK. The Atlantic Multidecadal Variability (AMV, Knight et al. 2006) and Atlantic Meridional Overturning Circulation (AMOC, Frajka-Williams et al. 2019) are thought to be the dominant modes of ocean variability in the North Atlantic on decadal timescales. However, variability in the AMOC and AMV is often found to be coherent (Zhang et al, 2019). Changes in ocean circulation result in changes in North Atlantic SST, however the impact of this variability on European climate may be larger in summer than winter (Zhange et al. 2019, and see below). However, there are questions in whether current climate models correctly capture the influence of variability in the North Atlantic Ocean on the atmospheric circulation. In particular, there is a signal-to-noise issue in decadal forecasts of the wintertime NAO that has recently been identified (Smith et al. 2020), which suggests that current climate models underestimate the magnitude of this influence. This will be discussed in more detail in section 4.2.

Summer: In summer, the North Atlantic and European atmospheric circulation tends to be more quiescent than in winter. In addition, the seasonality of the stratospheric circulation means that stratospheric variability is less important than in winter for driving variability in the troposphere. Recent studies have highlighted the role of the North Atlantic Ocean, and in particular decadal variations in sea surface temperature, such as the AMV, in modulating summer rainfall in NW Europe (Sutton and Dong, 2012, Ghosh et al. 2017). Other studies have suggested that interannual variability in North Atlantic SST might influence the summer atmospheric circulation and precipitation over the UK. This includes through subtropical Atlantic SSTs and Rossby waves (Wulff et al. 2017), or by spring North Atlantic SSTs influencing the summer evolution of the summer North Atlantic circulation (Osso et al. 2018, 2020).

There have been a number of recent papers which focus specifically on the impact of the rapid Arctic Sea ice loss on the summer mid-latitude circulation (Coumou et al. 2018). This includes i) through the weakening and/or shifts in the mid-latitude jet streams by weakening the equator-to-pole temperature gradient (e.g. Chang et al. 2016) and ii) through the amplification of circumpolar Rossby waves which may give rise to quasi-resonant circulation regimes (e.g. Coumou et al. 2014). Recent studies have also highlighted the formation of double jet streams in the context of European heatwaves (Rousi et al. 2022). Modelling studies from the CMIP5 climate model ensemble have suggested some evidence of a link between circumpolar Rossby wave guides and a reduction in Arctic Sea ice (Mann et al. 2017), but more recent modelling studies have highlighted the role of anthropogenic aerosol in driving these changes (Dong et al. 2022).

Given the number of hypotheses and conflicting lines of evidence, understanding the impact of Arctic Sea ice loss and warming on the mid-latitude circulation continues to be an area of active research.

The role of ENSO in driving North Atlantic and European atmospheric variability is thought to be less important in summer then in winter as i) ENSO tends to peak in boreal winter and is much weaker during summer (Yu et al. 2017), and ii) the prevailing easterly winds in the tropics in summer reduce the ability of Rossby waves to propagate into the extratropics (Schubert et al. 2011). Studies looking at relationship between ENSO and summer European climate therefore tend to find weaker relationships, for example, two recent studies found weak relationships that were nonstationary (Martija-Diez et al. 2021, 2022).

4.2. The potential for skilful subseasonal-to-decadal forecasts of low rainfall and drought in the UK

Skilful s2d forecasts for Europe would provide substantial socioeconomic benefits, e.g. for the management of water resources. Initial studies focused on the use of statistical models to generate skilful forecasts (e.g. Colman et al. 1999, Wang et al. 2017). Recent improvements in dynamical s2d forecasting systems, however, have led to skilful dynamical seasonal forecasts of the wintertime North Atlantic Oscillation (Scaife et al. 2014, Athanasiadis et al. 2017). Improvements in forecasts of the large scale atmospheric circulation have led to improved skill in seasonal forecasts of anticyclonic blocking over Europe (Davini et al., 2021) and UK and Ireland rainfall (Scaife et al 2014, Baker et al. 2018b, Stringer et al. 2022, Golian et al. 2022).

Despite the improved skill in wintertime seasonal forecasts of the NAO and associated impacts there are still substantial challenges. Firstly, wintertime seasonal forecasts of the NAO currently suffer from a signal-to noise issue (Eade et al., 2014, Baker et al. 2018a), where the predictive signal in forecasts is substantially smaller than that suggested from statistical analysis (Kumar, 2009). The signal-to-noise issue has a number of implications (e.g. that large ensemble of forecasts are required to assess predictive signals) and also suggests that large-scale drivers of predictability and/or dynamical processes are not correctly represented in current dynamical seasonal forecast systems. In addition, seasonal forecast skill in the NAO appears to be intermittent and varies substantially from decade to decade (Weisheimer et al. 2017). This produces sampling issues in terms of assessing skill, given the relatively short period over which dynamical seasonal forecasts can be initialised and evaluated.

Secondly, dynamical seasonal forecasts in other seasons have less skill than in winter. In particular, summer forecasts have very little skill for forecasting the atmospheric circulation over the North Atlantic and Europe, which may be due to dynamical forecast systems not capturing some key processes that are thought to be important (e.g., coupled ocean-atmosphere process in the North Atlantic ocean in spring and summer, Osso et al. 2000). It is worth noting that although there is little skill in atmospheric circulation, some dynamical seasonal forecast systems have reported modest skill in making summer forecasts of NW European summer precipitation (Dunstone et al. 2018) and land surface temperatures (Prodhomme et al. 2022), associated with longer timescale memory and thermal inertia in land surface temperatures and North Atlantic sea surface temperatures.

Skilful forecasts of the North Atlantic sea surface temperatures on multi-annual to decadal timescales have been developed in a number of decadal forecasting systems. As mentioned in section 4.1 , multi-annual to decadal variations in North Atlantic sea surface temperatures plays a role in modulating the climate of Europe and the UK (Zhang et al. 2019). However, the skill is more limited for decadal forecasts for variables over land (Smith et al. 2019). Recent studies have highlighted that the signal-to-noise issue may also be present in multi-annual to decadal forecasts of the North Atlantic atmospheric circulation (Smith et al. 2022). This raises the possibility that improvements in decadal forecasting systems may lead to skilful forecasts on longer timescales for North Atlantic atmospheric circulation and climate.

In summary, there have been recent improvements in s2d dynamical forecasts of the atmospheric circulation, rainfall and temperature for the UK and Europe. These improvements, coupled with new machine learning and artificial intelligence approaches that can produce skilful statistical forecasts of the NAO, atmospheric circulation, and rainfall (e.g. Wang et al. 2017, Hall et al. 2019), may provide a step change in our capacity to provide skilful s2d forecasts.

There are still substantial challenges in using such forecasts. In particular, the skill found in the atmospheric component of s2d forecasts needs to be combined with the additional skill found in hydrological forecast systems (Bell at al. 2017) in order to produce s2d forecasts useful for water resource managers. In addition, it is important to recognise the additional challenges and barriers to the uptake of s2d forecasts, e.g. the use of forecasts in risk averse drought management conditions (e.g. Lopez and Haines, 2017) and the translation of s2d forecasts into measures and terms of reference that are more relevant for drought managers. Nonetheless, recent improvements in both hydrological and atmospheric s2d forecasts suggest that further investment in s2d forecasting systems may yield substantial benefits for monitoring and early warning of drought.

5. Climate change, atmospheric circulation, and low rainfall and drought in the UK

In section 5, the impact of climate change on North Atlantic and European atmospheric circulation, low rainfall and drought in the UK will be considered. Assessing the impacts of climate change on phenomena such as the atmospheric circulation requires projections from climate models, since the phenomena are too complex to ascertain a priori or with simpler models. However, to have any confidence in climate models, it is essential that they are evaluated using all available observations to determine whether they are fit for purpose. Section 5.1 will therefore consider the biases in climate models for North Atlantic and European atmospheric circulation and rainfall. Sections 5.2 and 5.3 will overview climate model projections for winter and summer respectively, and section 5.4 will provide a summary. Section 5 will draw upon evidence from a range of climate model projections, including those from the Coupled Model Intercomparison Project (CMIP), which has been used to inform the IPCC assessment reports on climate change, and the UK Climate Projections (UKCP).

5.1. Climate model biases in North Atlantic and European atmospheric circulation and rainfall

In order to have any confidence in climate model projections, it is essential to evaluate their ability to represent the key processes of interest. For the North Atlantic and European atmospheric circulation and rainfall in the UK, this means evaluating the representation of key aspect of the circulation, such as the North Atlantic jet stream, and the drivers of drought, such as anticyclonic blocking.

The observed wintertime atmospheric circulation over Europe is characterised by the westerly jet stream over the North Atlantic. Storms, on average, travel across the North Atlantic ocean and recurve north of Scotland, bringing rainfall into the UK. In climate models, the position of the North Atlantic jet stream is biased southward in present-day simulations, with the result that, on average, storms travel over the UK and onwards into central Europe (Priestly et al. 2020). There has been a steady improvement in the position of the North Atlantic jet stream in climate models, from the CMIP3 models used to inform the Third IPCC assessment report on climate change (IPCC AR3 2001) through to the current CMIP6 models (Harvey et al. 2020). However, a few models in the CMIP6 ensemble still place the North Atlantic jet stream too far south.

Climate models also have problems representing processes such as anticyclonic blocking conditions, that give rise to prolonged periods of low rainfall and drought in the UK. Figure 5.1 shows the North Atlantic and European blocking frequencies from the CMIP5, CMIP6 historical simulations and ERA-5 reanalyses. Consistent with the wintertime North Atlantic jet stream being too far south, the CMIP6 climate models typically underestimate the frequency of winter anticyclonic blocking events over Europe and the UK by approximately 20% compared to the ERA-5 reanalysis (Davini et al. 2020, Schiemann et al. 2020). In summer, there is a similar underestimation in the frequency of anticyclonic blocking in the CMIP6 climate models (Figures 5.1).

There is some indication that the biases in blocking (Figure 5.1) and the position of the jet stream (e.g. Baker et al. 2019, not shown) are smaller in higher resolution climate models. This effect appears to be more important in summer than in winter, however, which suggests other processes (e.g. the representation of mountains, gravity waves, and ocean coupling) may also be important. In summary, the CMIP5 and CMIP6 biases in circulation are large, which has implications for assessing the impacts of climate change on low rainfall and drought in the UK. These implications will be discussed more fully in section 5.4.

A recent application of climate models and atmospheric forecasts is to use them generate long time series of synthetic weather that can be used to inform current and future risk assessments (sometimes referred to as the UNSEEN method, Thompson et al. 2017). These methods have long been used by the insurance industry in catastrophe models of hydro-meteorological hazards. Large ensembles of weather and seasonal forecasts have also been used to investigate extreme waves (Breivik et al. 2013), extreme precipitation (Thompson et al. 2017), precipitation deficits for maize production (Kent et al. 2017), and to detect decadal changes in precipitation extremes (Kelder et al. 2020). Large ensembles of climate model simulations have also been used to generate synthetic event sets of drought in the UK (Guillod et al. 2018).

It is essential to ensure that models used in these methods are capable of capturing key processes, which is challenging given the substantial model biases and short observational records. For UK droughts, this would include capturing both multi-annual droughts and the long-drought type events seen in the UK in the 19th Century. Recent research has evaluated European soil-moisture drought in climate models, motivated by the recent 2018-2020 European droughts (Rakovec et al. 2022). However, there is a substantial research gap in terms of assessing the climate model representation of drought on longer timescales and in UK contexts. Despite these challenges, there may be new opportunities to investigate these methods using single-model, large-ensemble simulations, which produce long continuous simulations (Deser et al. 2020).

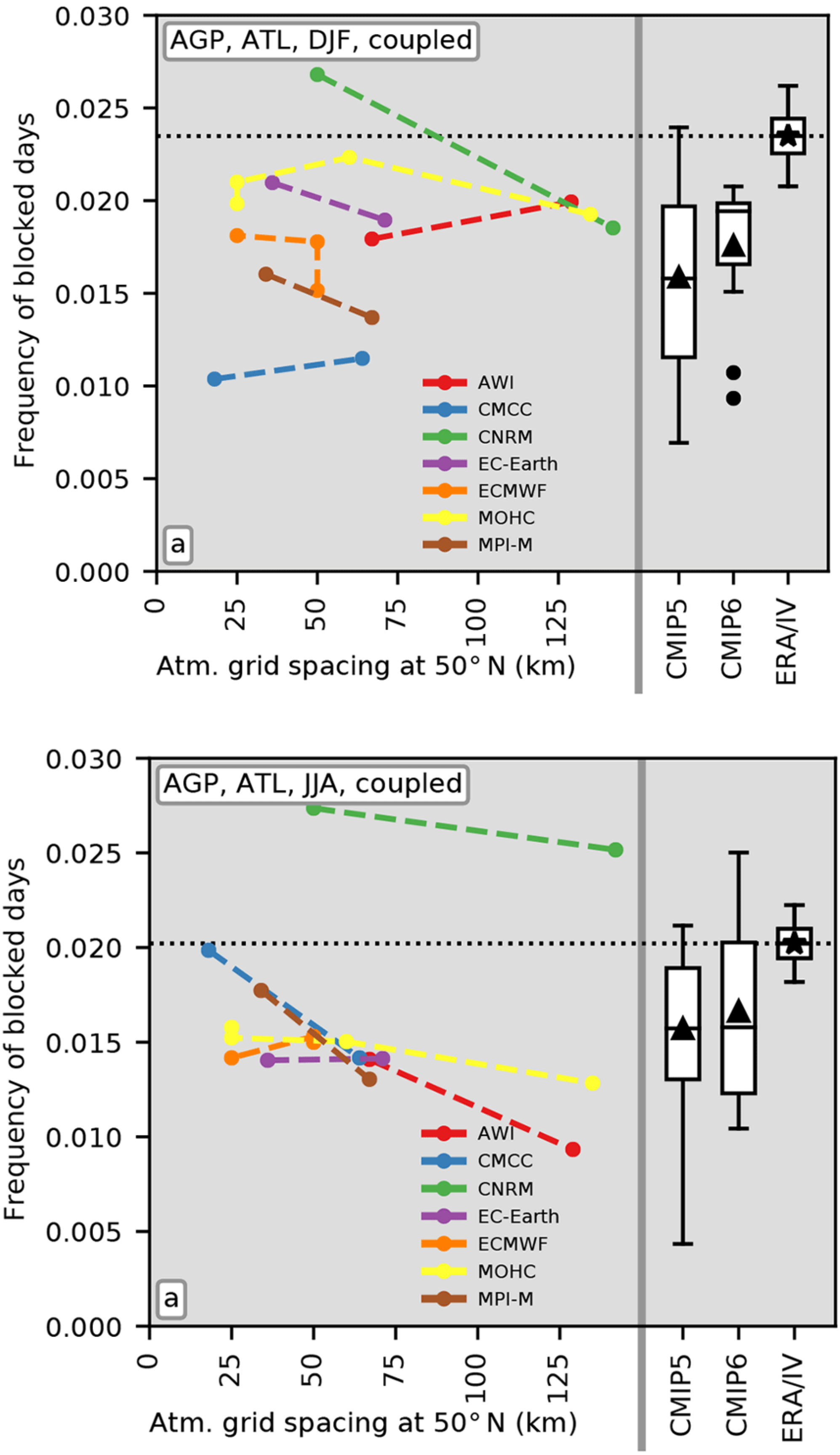

Figure 5.1: Frequency of anticyclonic blocking over the North Atlantic and Europe in the CMIP5, CMIP 6 and ERA5 reanalyses for left) DJF and right) JJA. Colours, same but for the Hiresmip high-resolution climate model ensemble and are plotted as a function of grid spacing in the atmospheric component of the coupled model to

highlight the relationship between model resolution and representation of blocking. From Schiemann et al. (2020).

Although the focus of this essay is on the atmospheric circulation and low rainfall, it is worth noting other biases in climate models that are relevant for drought. For example, climate models tend to underestimate non-local negative soil-moisture–precipitation feedbacks. Taylor et al. (2013) found that climate models with parameterised convection tend to underestimate the local circulation induced by gradients in surface moisture, which has important consequences for the evolution of soil moisture in climate projections. Although these biases are more important for tropical projections of climate change, they should be considered in wider assessments of future drought.

5.2. Climate change and impacts on wintertime rainfall and atmospheric circulation in the UK

Climate change could have a number of impacts on precipitation over the UK. Firstly, the midlatitude atmospheric circulation could change (often referred to as the dynamical response), which will alter transports of moisture and potentially impact precipitation. Secondly, the warmer atmosphere will be able to hold more moisture (often referred to as thermodynamic response) which again could impact precipitation. The overall change precipitation will largely be in response to these two factors. Although the focus of this essay is on atmospheric circulation and precipitation, it should be noted that wider assessments of the impact of climate change on drought need to also consider changes in evapotranspiration. Broadly, increases in land temperature will also tend to increase evaporation, and the total evapotranspiration may modified further by the response of vegetation.

As the climate warms in climate projections, the atmospheric circulation responds by shifting the zonally-averaged jet stream polewards (IPCC AR6, 2021). This is in response to enhanced warming in tropical upper troposphere, which increases the upper tropospheric equator-to-pole temperature gradient, and enhanced warming over the Arctic, which reduces the lower tropospheric equator-to-pole temperature gradient. These changes affect the position and strength of the jet stream through thermal wind balance (Shaw et al. 2016).

Over the wintertime North Atlantic ocean, however, the response of the atmospheric circulation in climate models is very different. Changes in North Atlantic ocean circulation result in a strengthening of the lower tropospheric equator-to-pole temperature gradient. The overall effect is an extension of the surface westerlies and jet stream over the North Atlantic into NW Europe (Ulrich et al. 2008, Zappa et al 2013, Harvey et al. 2020). This is indicated in Figure 5.2c, which shows the lower tropospheric jet stream strength over NW Europe increasing in time in the CMIP6 models (Harvey et al. 2023). Consistent with the extension of the North Atlantic jet stream, wintertime European blocking becomes less frequent and persistent in climate model projections (Masato et al. 2013, Kitano et al. 2016, Lee et al. 2017, Matsueda et al. 2017, Woollings et al. 2018, Davini et al. 2020). From a weather regime perspective, there is a decrease in the persistent and frequency of weather regimes associated with European blocking (Dorrington et al. 2022).

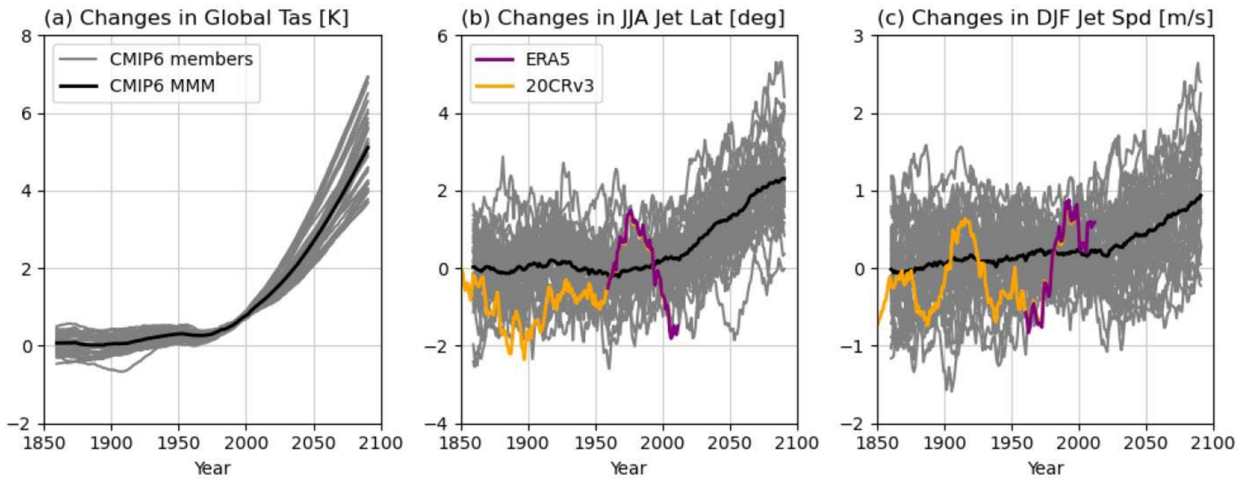

Figure 5.2: CMIP6 projections of changes in (left) global mean temperature (centre) JJA jet latitude and (right) DJF jet speed. All values are 20-year running mean anomalies relative to a quasi-preindustrial value, as described in Harvey et al. (2023). All panels show (grey) the 38 CMIP6 models together with (black) their multi-model mean and (purple, orange) the observed time series from ERA5 and 20CRv3, respectively. From Harvey et al. (2023).

Nearly all climate models agree on the sign of the responses in the wintertime North Atlantic and European atmospheric circulation, however, there is a substantial spread in the magnitude of the responses. The large spread of responses, combined with the relatively large atmospheric circulation biases seen in climate models, and the difficulty in detecting trends in atmospheric circulation (section 2.3) led the IPCC AR6 to conclude that overall there is “low confidence in the effects of greenhouse gas forcing on changes in atmospheric dynamics” (IPCC AR6, 2021).

The thermodynamic response of atmospheric moisture and precipitation to climate change is predicated on the Clausius-Clapeyron equation. Increases in wintertime precipitation in the extratropics and Northern Europe in response to climate change are robustly found in climate models. Consequently, there is more confidence in the thermodynamic responses and drivers of drought in response to climate change than the dynamical responses (IPCC AR6, 2021).

Assessed from a perspective of drought metrics such as 6-month SPI (Standardised Precipitation Index), Northern European meteorological drought is projected to decrease in the future associated with increasing winter precipitation, especially for the UK and Ireland (Touma et al. 2015; Spinoni et al., 2020; Vicente-Serrano et al. 2021). As noted in section 3.2, trends in SPI can disagree with other drought indices that include evapotranspiration (Stagge et al. 2017).

5.3. Climate change and impacts on atmospheric circulation and summertime rainfall in the UK

The response of the North Atlantic and European atmospheric circulation to climate change has a different character in summer to that in winter. Climate models project that the North Atlantic jet stream shifts poleward in summertime. The poleward shift in the North Atlantic jet stream is seen robustly in all CMIP6 climate model projections although, similar to winter, the magnitude in the spread of responses is large (Figure 5.2b, Harvey et al. 2023). The poleward shift in the summer North Atlantic jet stream is consistent with the changes expected from the enhanced warming in the upper tropical troposphere and the strengthening of the equator-to-pole temperature gradient (Shaw et al. 2016). The response of the atmospheric circulation to changes in the North Atlantic ocean is substantially weaker, since surface fluxes into the atmosphere surface ocean are substantially smaller in summer than in winter.

Interpreting these circulation changes in terms of current patterns of variability, the poleward migration of the summer jet stream is consistent with more positive SNAO (Murphy et al. 2018). A similar result was found for the weather types associated with dry UK summers in the UKCP18 simulations (Pope et al. 2022). In similar fashion, De Luca et al. (2019) also found that persistence of Lamb Weather Types were projected to increase in summer. The northwards migration of the North Atlantic jet stream is accompanied by the slight reduction in anticyclonic blocking over NW Europe and a slight increase further over Scandinavia (Masato et al. 2013, Woollings et al. 2018, Davini et al. 2020). However the details of projected changes in summertime blocking are sensitive to the exact blocking index employed.

UK Rainfall in summer is projected to decrease in summer (IPCC AR6 2021). The UKCP18 projections suggest summer rainfall could decrease more strongly in England than in Scotland (UKCP18, 2019). A reduction in rainfall implies that changes in circulation must be reducing moisture transports into the UK, given warmer summer air temperatures will able to hold more moisture. Murphy et al. (2018) found a strong relationship between the projected decrease in summer UK rainfall in the CMIP5 and UKCP18 models and a projected increase in the SNAO. The nature of summer rainfall is also projected to change substantially, with convective-permitting climate model simulation suggesting summer rainfall could occur less frequently but be more intense (Kendon et al. 2014). Similar to winter, a large spread in the projected decrease of summer rainfall in the UK and NW Europe is seen in the UKCP18 and CMIP6 models.

The projected decrease in summer rainfall from climate projections is not consistent with the observed changes seen over the past dew decades, which show a very slight increase in rainfall (Figure 3.1; Kendon et al. 2022). An additional conclusion from Figure 5.2c is that the summer North Atlantic jet has shifted southwards over the past few decades, which is consistent with the positive signal seen in the SNAO and increased UK rainfall. This southwards shift is i) large compared to the observed variability of the North Atlantic jet stream and large compared to the variability of the CMIP6 climate models and ii) the shift is opposite to the future direction projected from climate models. The extent to which recent trends in North Atlantic atmospheric circulation, and associated impacts on UK rainfall, are due to external forcing (greenhouse gases, anthropogenic aerosol, etc..) versus internal variability are still not clear. This remains a major knowledge gap in terms of assessing drought risk in the UK over the next few decades, since it is plausible that the drying signal in UK summer rainfall projected from climate models may be ‘masked’ in observations by the internal variability of the North Atlantic atmospheric circulation.

5.4. Summary

The overall picture suggested by climate model projections is warmer, wetter winters and hotter, drier summers for the UK (UKCP18 science overview report, 2019). In winter, this is accompanied by an extension of the North Atlantic jet stream and westerlies into NW Europe, increased storminess, and a reduction in anticyclonic blocking. In summer, the North Atlantic jet stream is projected to shift northwards. There is a large spread in projections in both the UKCP18, CMIP5, and CMIP6 models.

The changing nature in the seasonality of rainfall has a number of implications in terms of drought, with wetter winters decreasing the probability of drought and drier summers enhancing it. In addition, the discussion in this essay has focused on atmospheric circulation and precipitation, however, the role played by evapotranspiration could become more important in the future as temperatures warm, and be exacerbated by land-surface feedbacks linked to soil moisture deficits and sensible heating. The nature of drought may therefore change, for example an increasing incidence of flash droughts (Shah et al. 2022). In addition, other aspects of hydrological cycle may need to be considered, for example, drought termination (Parry et al. 2016) and the increasingly rapid seesaw between drought and flood conditions (He and Sheffield, 2020).

A key consideration is how to assess future drought risks in the UK, especially given the discussion above regarding the large spread in the projected responses of the atmospheric circulation and precipitation over the North Atlantic and Europe. It is also essential to consider the biases in climate models, for example the underestimation of blocking frequency, when assessing the regional impacts of climate change on rainfall and drought in the UK. One way forward to addressing these large uncertainties is to adopt “Storyline” approaches (Shepherd et al. 2018), which develop physically self-consistent scenarios, especially for plausible future events or pathways. These approaches have started to be applied to UK droughts (Chan et al. 2022) and the North Atlantic jet stream (Harvey et al. 2023) and could provide valuable insight into the assessment of the impacts of climate change on UK drought.

6. Conclusions and research questions

In section 6, the key conclusions and five potential research questions identified in this essay are summarised.

How has the occurrence of low rainfall and drought in the UK varied in the historical record? Historical records enable a view of UK drought over two and a half centuries. During that time, the

UK has experienced lengthy periods of low rainfall, including the 'long droughts' in the 19th Century (1854-1860, and 1890-1910). Addressing these issues will require investment in recovering old weather records and developing long-term atmospheric reanalysis. In addition, the reanalysis of the past needs to be combined with multiple lines of evidence (meteorological, hydrological, impacts) to provide the best assessment of historical drought. Currently, we have a poor understanding of the atmospheric drivers of the long droughts, which raises questions whether we fully understand the risk of such events were they to occur today.

1. What role does North Atlantic and European multidecadal climate variability play in UK drought risk, especially for producing events similar to the historical long droughts?

What are the prospects for improving early warnings of drought on s2d (subseasonal-to-decadal) timescales? There have been recent improvements in s2d dynamical forecasts of the atmospheric circulation, for example, skilful dynamical seasonal forecasts of the wintertime North Atlantic Oscillation are now possible. However, there remain substantial challenges and barriers to the uptake of s2d forecasts by water resource managers. A key question is whether further improvements could be obtained by coupling s2d forecasts of the North Atlantic atmospheric circulation with hydrological forecasts and new machine learning approaches.

2. When coupled with hydrological forecasts and new machine learning approaches, to what extent could the recent improvements in dynamical s2d forecasts provide useful forecasts of the North Atlantic atmospheric circulation and UK rainfall for the early warning of drought?

What impact could climate change have on the occurrence of low rainfall and drought in the UK, and associated variability in atmospheric circulation? Climate change is expected to make UK winters warmer and wetter and summers hotter and drier. The increase in rainfall in winter is partly associated with warmer temperatures and thus more moisture and rainfall, and partly associated with an extension of the North Atlantic jet stream over the UK and thus more storms. In summer, the projected decrease in rainfall is associated with the northwards shift in the jet stream, which leads to a reduction in rainfall. In addition, hotter summer temperatures could be associated with an increase in evaporation.

The projected climate model decrease in summer rainfall does not agree with observations, which show a very slight increase in rainfall over the past few decades. Over the same time period there has been a marked southwards shift in the observed position of the summertime North Atlantic jet stream, which may partly explain the increasing trend seen in observed UK rainfall. The recent southwards shift in the summer North Atlantic jet, however, is opposite to the northwards shift expected from climate model projections. At present, it remains unclear whether recent trends in North Atlantic atmospheric circulation, and their associated impacts on UK rainfall, are due to external forcing (greenhouse gases, anthropogenic aerosol, etc..) or internal variability. It is plausible that the small increases in summer rainfall observed over the past few decades may be due to internal climate variability and might be ‘masking’ the expected drying trend found in climate models. If so, this has severe implications for the assessment of UK drought risk over the next few decades.

3. To what extent are the recent trends in summer atmospheric circulation, and associated impacts on UK rainfall, due to internal variability? If so, are they potentially opposing the trend we might expect from climate change?

There is also substantial uncertainty in climate model projections, which in part may be due to climate model biases in atmospheric circulation, such as the underestimation of the frequency of blocking patterns. Thus, there is a need to improve climate models and the representation of key processes. For phenomena directly relevant to drought, this includes improving the representation of anticyclonic blocking, atmospheric eddy-mean flow feedbacks, clouds and aerosols, boundary layers, land-surface feedbacks such as soil drying and vegetation water stress. There is evidence, however, that higher resolution climate models may help reduce atmospheric circulation biases. In addition, approaches such as Storylines could be employed to help gain insight into future UK drought risk. Despite these biases, large ensembles of climate model simulations might be helpful in generating event sets of modelled droughts to help inform current and future drought risk assessments, especially the 1-in-500 year events required for planning in water resource management.

4. Is there a role for large ensembles of climate model simulations be used to generate synthetic event sets of droughts to help inform current and future drought risk assessments? Or, are model biases still too large?

5. Given these large uncertainties should additional approaches be employed to evaluate future UK drought risk, for example, storyline approaches?

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References

Alexander, L.V. and Jones, P.D. (2001) Updated precipitation series for the U.K. and discussion of recent extremes, Atmospheric Science Letters doi:10.1006/asle.2001.0025.

Allan, R., Brohan, P., Compo, G. P., Stone, R., Luterbacher, J., & Brönnimann, S. (2011). The International Atmospheric Circulation Reconstructions over the Earth (ACRE) Initiative, Bulletin of the American Meteorological Society, 92(11), 1421-1425.

Alley, W.M., (1984): The Palmer Drought Severity Index: limitations and assumptions. Journal of Applied Meteorology, 23:1100–1109.

Ambaum, M. H. P., Hoskins, B. J. and Stephenson, D. B. (2001) Arctic oscillation or North Atlantic oscillation? Journal of Climate, 14 (16). pp. 3495-3507. ISSN 1520-0442 doi: https://doi.org/10.1175/1520-0442(2001)014<3495:AOONAO>2.0.CO;2

Athanasiadis PJ, Bellucci A, Scaife AA, Hermanson L, Materia S, Sanna A, Borrelli A, MacLachlan C, Gualdi S. (2017) A Multisystem View of Wintertime NAO Seasonal Predictions, JOURNAL OF CLIMATE, volume 30, no. 4, pages 1461-1475, DOI:10.1175/JCLI-D-16-0153.1.

Andrews, M. B., Knight, J. R., Scaife, A. A., Lu, Y., Wu, T., Gray, L. J., & Schenzinger, V. (2019). Observed and simulated teleconnections between the stratospheric Quasi-Biennial Oscillation and Northern Hemisphere winter atmospheric circulation. Journal of Geophysical Research: Atmospheres, 124, 1219– 1232. https://doi.org/10.1029/2018JD029368

Barnes EA, Slingo J, Woollings T (2012) A methodology for the comparison of blocking climatologies across indices, models and climate scenarios. Clim Dyn 38(11–12):2467–2481

Barnes, E. A. (2013) Revisiting the evidence linking Arctic amplification to extreme weather in midlatitudes. Geophys. Res. Lett. 40, 4734–4739.

Barnes, E. A., Dunn-Sigouin, E., Masato, G., and Woollings, T. (2014), Exploring recent trends in Northern Hemisphere blocking, Geophys. Res. Lett., 41, 638– 644, doi:10.1002/2013GL058745.

Barnes EA, Screen JA (2015) The impact of Arctic warming on the midlatitude jet-stream: Can it? Has it? Will it? Wiley Interdiscip Rev Clim Change 6(3):277–286.

Baker, L. H., Shaffrey, L. C., Sutton, R. T., Weisheimer, A. and Scaife, A. A. (2018a) An intercomparison of skill and overconfidence/underconfidence of the wintertime North Atlantic Oscillation in multimodel seasonal forecasts. Geophysical Research Letters, 45 (15). pp. 7808-7817. ISSN 0094-8276 doi: https://doi.org/10.1029/2018GL078838

Baker, L., Shaffrey, L. and Scaife, A. A. (2018b) Improved seasonal prediction of UK regional precipitation using atmospheric circulation. International Journal of Climatology, 38 (S1). e437-e453. ISSN 0899-8418 doi: https://doi.org/10.1002/joc.5382

Baker, A. J., Schiemann, R., Hodges, K. I., Demory, M.-E., Mizielinski, M. S., Roberts, M. J., Shaffrey, L. C., Strachan, J. and Vidale, P. L. (2019) Enhanced climate change response of wintertime North Atlantic circulation, cyclonic activity and precipitation in a 25 km-resolution global atmospheric model. Journal of Climate, 32 (22). pp. 7763-7781. ISSN 1520-0442 doi: https://doi.org/10.1175/JCLI-D-19-0054.1

Barker, L. J., Hannaford, J., Chiverton, A., and Svensson, C. (2016) From meteorological to hydrological drought using standardised indicators, Hydrol. Earth Syst. Sci.,1 20, 2483–2505, https://doi.org/10.5194/hess-20-2483-2016

Barker, L. J., Hannaford, J., Parry, S., Smith, K. A., Tanguy, M., and Prudhomme, C. (2019) Historic hydrological droughts 1891–2015: systematic characterisation for a diverse set of catchments across the UK, Hydrol. Earth Syst. Sci., 23, 4583–4602, https://doi.org/10.5194/hess-23-4583-2019.

Baldwin, M.P. and Dunkerton, T.J. (2001) Stratospheric Harbingers of Anomalous Weather Regimes. Science, 294, 581-584 http:/dx.doi.org/10.1126/science.1063315

Bell, V A, Davies, H N, Kay, A L et al. (2017). A national-scale seasonal hydrological forecast system: development and evaluation over Britain. HESS. doi:10.5194/hess-21-4681-2017

Bellomo, K., Angeloni, M., Corti, S. et al. (2021) Future climate change shaped by inter-model differences in Atlantic meridional overturning circulation response. Nat Commun 12, 3659. https://doi.org/10.1038/s41467-021-24015-w

Bloomfield, H. C., Shaffrey, L. C., Hodges, K. I. and Vidale, P. L. (2018) A critical assessment of the long term changes in the wintertime surface Arctic Oscillation and Northern Hemisphere storminess in the ERA20C reanalysis. Environmental Research Letters, 13 (9). 094004. ISSN 1748-9326

Breivik, Ø., Aarnes, O. J., Bidlot, J.-R., Carrasco, A. & Saetra, Ø. (2013) Wave extremes in the northeast Atlantic from ensemble forecasts. J. Clim. 26, 7525–7540.

Cassou, C. (2008) Intraseasonal interaction between the Madden–Julian Oscillation and the North Atlantic Oscillation. Nature 455, 523–527. https://doi.org/10.1038/nature07286

Chan, W. C. H., Shepherd, T. G., Facer-Childs, K., Darch, G., and Arnell, N. W. (2022) Storylines of UK drought based on the 2010–2012 event, Hydrol. Earth Syst. Sci., 26, 1755–1777, https://doi.org/10.5194/hess-26-1755-2022.

Chang, E. K. M., Ma, C., Zheng, C. & Yau, A. M. W. (2016) Observed and projected decrease in Northern Hemisphere extratropical cyclone activity in summer and its impacts on maximum temperature. Geophys. Res. Lett. 43, 2200–2208.

Cohen, J., Zhang, X., Francis, J. et al. (2020) Divergent consensuses on Arctic amplification influence on midlatitude severe winter weather. Nat. Clim. Chang. 10, 20–29. https://doi.org/10.1038/s41558-019-0662-y

Colman, A. and Davey, M. (1999), Prediction of summer temperature, rainfall and pressure in Europe from preceding winter North Atlantic Ocean temperature. Int. J. Climatol., 19: 513-536. https://doi.org/10.1002/(SICI)1097-0088(199904)19:5<513::AID-JOC370>3.0.CO;2-D

Compo, G.P., J.S. Whitaker, P.D. Sardeshmukh, N. Matsui, R.J. Allan, X. Yin, B.E. Gleason, R.S. Vose, G. Rutledge, P. Bessemoulin, S. Brönnimann, M. Brunet, R.I. Crouthamel, A.N. Grant, P.Y. Groisman, P.D. Jones, M. Kruk, A.C. Kruger, G.J. Marshall, M. Maugeri, H.Y. Mok, Ø. Nordli, T.F. Ross, R.M. Trigo, X.L. Wang, S.D. Woodruff, and S.J. Worley, (2011) The Twentieth Century Reanalysis Project. Quarterly J. Roy. Meteorol. Soc., 137, 1-28. http://dx.doi.org/10.1002/qj.776

Cornes, R.C.; Jones, P.D.; Briffa, K.R.; Osborn, T.J. Estimates of the North Atlantic Oscillation back to 1692 using a Paris-London westerly index. (2013) Int. J. Climatol., 33, 228–248

Corti S., F. Molteni and T. N. Palmer (1999) Signature of recent climate change in frequencies of natural atmospheric circulation regimes. Nature 398, 799-802

Cotterill, D.F., Pope, J.O. & Stott, P.A. Future extension of the UK summer and its impact on autumn precipitation. Clim Dyn (2022). https://doi.org/10.1007/s00382-022-06403-0

Coumou D, Petoukhov V, Rahmstorf S, Petri S, Schellhuber H-J (2014) Quasi-resonant circulation regimes and hemispheric synchronization of extreme weather in boreal summer. Proc Natl Acad Sci USA 111(34):12331–12336. doi:10.1073/pnas.1412797111

Coumou, D., Di Capua, G., Vavrus, S. et al. (2018) The influence of Arctic amplification on mid-latitude summer circulation. Nat Commun 9, 2959. https://doi.org/10.1038/s41467-018-05256-8

Davini, P., & D’Andrea, F. (2020). From CMIP3 to CMIP6: Northern Hemisphere Atmospheric Blocking Simulation in Present and Future Climate, Journal of Climate, 33(23), 10021-10038.

Dayrell, Carmen; Svensson, Cecilia; Hannaford, Jamie ; McEnery, Tony; Barker, Lucy J. ; Baker, Helen; Tanguy, Maliko. (2022) Representation of drought events in the UK: contrasting 200 years of news texts and rainfall records. Frontiers in Environmental Science, 10, 760147. 18, pp. https://doi.org/10.3389/fenvs.2022.760147

Dawson, A., T. N. Palmer, and S. Corti (2012), Simulating regime structures in weather and climate prediction models, Geophys. Res. Lett., 39, L21805, doi:10.1029/2012GL053284

Davini, P, Weisheimer, A, Balmaseda, M, et al. (2021) The representation of winter Northern Hemisphere atmospheric blocking in ECMWF seasonal prediction systems. Q J R Meteorol Soc.; 147: 1344– 1363. https://doi.org/10.1002/qj.3974

De Luca, P., et al., (2019). Past and projected weather pattern persistence with associated multi-hazards in the British Isles. Atmosphere, 10, p.577.

Deser, C., F. Lehner, K. B. Rodgers, T. Ault, T. L. Delworth, P. N. DiNezio, A. Fiore, C. Frankignoul, J. C. Fyfe, D. E. Horton, J. E. Kay, R. Knutti, N. S. Lovenduski, J. Marotzke, K. A. McKinnon, S. Minobe, J.

Randerson, J. A. Screen, I. R. Simpson and M. Ting, (2020) Insights from Earth system model initial-condition large ensembles and future prospects. Nature Climate Change, doi: 10.1038/s41558-020-0731-2

Dong, B. , Sutton, R. T. , Shaffrey, L. and Harvey, B. (2022) Recent decadal weakening of the summer Eurasian westerly jet attributable to anthropogenic aerosol emissions. Nature Communications, 13. 1148. ISSN 2041-1723 doi: https://doi.org/10.1038/s41467-022-28816-5

Dörnbrack, A., Pitts, M. C., Poole, L. R., Orsolini, Y. J., Nishii, K., and Nakamura, H. (2012). The 2009–2010 Arctic stratospheric winter – general evolution, mountain waves and predictability of an operational weather forecast model, Atmos. Chem. Phys., 12, 3659–3675, https://doi.org/10.5194/acp-12-3659-2012.

Dorrington, J., Strommen, K., Fabiano, F., & Molteni, F. (2022). CMIP6 models trend toward less persistent European blocking regimes in a warming climate. Geophysical Research Letters, 49, e2022GL100811. https://doi.org/10.1029/2022GL100811

Eade R, Smith D, Scaife A, Wallace E, Dunstone N, Hermanson L, Robinson N. (2014). Do seasonal-to-decadal climate predictions underestimate the predictability of the real world? Geophys. Res. Lett. 41(15): 5620– 5628. https://doi.org/10.1002/2014GL061146.

Folland, C. K., Knight, J., Linderholm, H. W., Fereday, D., Ineson, S., & Hurrell, J. W. (2009). The Summer North Atlantic Oscillation: Past, Present, and Future, Journal of Climate, 22(5), 1082-1103.

Folland, C. K., Hannaford, J., Bloomfield, J. P., Kendon, M., Svensson, C., Marchant, B. P., Prior, J., and Wallace, E. (2015). Multi-annual droughts in the English Lowlands: a review of their characteristics and climate drivers in the winter half-year, Hydrol. Earth Syst. Sci., 19, 2353–2375, https://doi.org/10.5194/hess-19-2353-2015, 2015.

Francis, J. A. & Vavrus, S. J. (2012) Evidence linking Arctic amplification to extreme weather in mid-latitudes. Geophys. Res. Lett. 39, https://doi.org/10.1029/2012GL051000. Influential early observational study arguing that Arctic amplification is contributing to more extreme weather in all seasons.

Frajka-Williams, E. and others, (2019), Atlantic Meridional Overturning Circulation: Observed Transport and Variability, Frontiers in Marine Science, Vol 6, https://doi.org/10.3389/fmars.2019.00260

Ghosh, R.WA Müller, J Baehr, J Bader, (2017) Impact of observed North Atlantic multidecadal variations to European summer climate: A linear baroclinic response to surface heating. Clim Dyn 48, 3547–3563.

Golian S, Murphy C, Wilby RL, Matthews T, Donegan S, Quinn DF, Harrigan S. (2022) Dynamical-statistical seasonal forecasts of winter and summer precipitation for the Island of Ireland. Int J Climatol. Sep;42(11):5714-5731. doi: 10.1002/joc.7557. Epub 2022 Feb 15. PMID: 36245684; PMCID: PMC9540122.

Guillod, B. P., Jones, R. G., Dadson, S. J., Coxon, G., Bussi, G., Freer, J., Kay, A. L., Massey, N. R., Sparrow, S. N., Wallom, D. C. H., Allen, M. R., and Hall, J. W. (2018) A large set of potential past, present and future hydro-meteorological time series for the UK, Hydrol. Earth Syst. Sci., 22, 611–634, https://doi.org/10.5194/hess-22-611-2018.

Hall, R.J., Wei, H. and Hanna, Edward (2019) Complex Systems Modelling for Statistical Forecasting of Winter North Atlantic Atmospheric Variability: a New Approach. Quarterly Journal of the Royal Meteorological Society, 145 (723). pp. 2568-2585. ISSN 0035-9009

Hall, R.J. and Hanna, E. (2018), North Atlantic circulation indices: links with summer and winter UK temperature and precipitation and implications for seasonal forecasting. Int. J. Climatol, 38: e660-e677. https://doi.org/10.1002/joc.5398

Hanna, Edward, Thomas E. Cropper, Richard J. Hall, Richard C. Cornes, and Mariano Barriendos. (2022) Extended North Atlantic Oscillation and Greenland Blocking Indices 1800–2020 from New Meteorological Reanalysis, Atmosphere 13, no. 3: 436. https://doi.org/10.3390/atmos13030436

Hannaford, J, Lucy Barker, Stephen Turner, Simon Parry, Maliko Tanguy, Amulya Chevuturi, (2023), Past Variability in Observed Hydrological River Flow Drought, Report to EA.

Hardiman, SC, Dunstone, NJ, Scaife, AA, et al. (2020) Predictability of European winter 2019/20: Indian Ocean dipole impacts on the NAO. Atmos Sci Lett.; 21:e1005. https://doi.org/10.1002/asl.1005

Harvey, B. J., Cook, P., Shaffrey, L. C., & Schiemann, R. (2020). The response of the northern hemisphere storm tracks and jet streams to climate change in the CMIP3, CMIP5, and CMIP6 climate models. Journal of Geophysical Research: Atmospheres, 125, e2020JD032701. https://doi.org/10.1029/2020JD032701

Harvey, B., Hawkins, E. and Sutton R. T. (2023) Storylines for future changes of the North Atlantic jet and associated impacts on the UK, (In review).

Hawcroft, M., Shaffrey, L., Hodges, K. and Dacre, H. (2012) How much Northern Hemisphere precipitation is associated with extratropical cyclones? Geophysical Research Letters, 39. ISSN 0094-8276

He, X., & Sheffield, J. (2020). Lagged compound occurrence of droughts and pluvials globally over the past seven decades. Geophysical Research Letters, 47, e2020GL087924. https://doi.org/10.1029/2020GL087924

Hollis, D, McCarthy, MP, Kendon, M, Legg, T, Simpson, I. (2019) HadUK-Grid—A new UK dataset of gridded climate observations. Geosci Data J.; 6: 151– 159. https://doi.org/10.1002/gdj3.78

Honda, M., Inoue, J. & Yamane, S. (2009) Influence of low Arctic sea-ice minima on anomalously cold Eurasian winters. Geophys. Res. Lett. 36, https://doi.org/10.1029/2008GL037079.

Hoskins, B. J., & Karoly, D. J. (1981). The Steady Linear Response of a Spherical Atmosphere to Thermal and Orographic Forcing, Journal of Atmospheric Sciences, 38(6), 1179-1196.

Hoy, A, Hänsel, S, Maugeri, M. (2020) An endless summer: 2018 heat episodes in Europe in the context of secular temperature variability and change. Int J Climatol.; 40: 6315–6336. https://doi.org/10.1002/joc.6582

Hurrell (1995) Decadal Trends in the North Atlantic Oscillation: Regional Temperatures and Precipitation, Science, 269, 676, DOI: 10.1126/science.269.5224.676

Ineson S, Scaife AA. (2009) The role of the stratosphere in the European climate response to El Nino, NATURE GEOSCIENCE, volume 2, no. 1, pages 32-36, DOI:10.1038/NGEO381.

IPCC, (2001): Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change [Houghton, J.T., Y. Ding, D.J. Griggs, M. Noguer, P.J. van der Linden, X. Dai, K. Maskell, and C.A. Johnson (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 881pp

IPCC, (2021): Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Masson-Delmotte, V., P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T.K. Maycock, T. Waterfield, O. Yelekçi, R. Yu, and B. Zhou (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 2391 pp. doi:10.1017/9781009157896.

Jones, P.D., Hulme, M. and Briffa, K.R. (1993), A comparison of Lamb circulation types with an objective classification scheme. Int. J. Climatol., 13: 655-663. https://doi.org/10.1002/joc.3370130606 Jones, P.D. and Conway, D. (1997a) Precipitation in the British Isles: an analysis of area-average data updated to 1995. International Journal of Climatology, 17, 427–438

Jones, P.D., Jonsson, T. and Wheeler, D. (1997b), Extension to the North Atlantic oscillation using early instrumental pressure observations from Gibraltar and south-west Iceland. Int. J. Climatol., 17: 1433-1450. https://doi.org/10.1002/(SICI)1097-0088(19971115)17:13<1433::AID-JOC203>3.0.CO;2-P

Karl, T. R., Nicholls, N., and Ghazi, A. (1999): CLIVAR/GCOS/WMO workshop on indices and indicators for climate extremes Workshop summary, Climatic Change, 42, 3–7.

Kelder, T., Müller, M., Slater, L.J. et al. Using UNSEEN trends to detect decadal changes in 100-year precipitation extremes. (2020) npj Clim Atmos Sci 3, 47. https://doi.org/10.1038/s41612-020-00149-4

Kendon, E., Roberts, N., Fowler, H. et al. Heavier summer downpours with climate change revealed by weather forecast resolution model. (2014) Nature Clim Change 4, 570–576. https://doi.org/10.1038/nclimate2258

Kendon, M., McCarthy, M., Jevrejeva, S., Matthews, A., Sparks, T., Garforth, J., & Kennedy, J. (2022). State of the UK Climate 2021. International Journal of Climatology, 42(Suppl. 1)(S1), 1– 80. https://doi.org/10.1002/joc.7787

Kent, C. et al (2017) Using climate model simulations to assess the current climate risk to maize production. Environ. Res. Lett. 12 054012

Kitano Y, Yamada TJ. (2016) Relationship between atmospheric blocking and cold day extremes in current and RCP8. 5 future climate conditions over Japan and the surrounding area. Atmos Sci Lett.;17(11):616–22.

Knight, J. R., Folland, C. K., and Scaife, A. A. (2006), Climate impacts of the Atlantic Multidecadal Oscillation, Geophys. Res. Lett., 33, L17706, doi:10.1029/2006GL026242.

Krueger, O., Schenk, F., Feser, F., & Weisse, R. (2013). Inconsistencies between Long-Term Trends in Storminess Derived from the 20CR Reanalysis and Observations, Journal of Climate, 26(3), 868-874.

Krueger, O., Feser, F., & Weisse, R. (2019). Northeast Atlantic Storm Activity and Its Uncertainty from the Late Nineteenth to the Twenty-First Century, Journal of Climate, 32(6), 1919-1931.

Lamb, H.H. (1950) Types and Spells of Weather around the Year in the British Isles. Quarterly Journal of the Royal Meteorological Society, 76, 393-438.

Lavers D, Prudhomme C, Hannah DM. (2010). Large-scale climate, precipitation and British River flows: identifying hydroclimatological connections and dynamics. J. Hydrol. 395(3): 242– 255. https://doi.org/10.1016/j.jhydrol.2010.10.036.

Lee DY, Ahn JB. (2017) Future change in the frequency and intensity of wintertime North Pacific blocking in CMIP5 models. Int J Climatol.;37(5):2765–81.

Lloyd-Hughes, B. and Saunders, M.A., (2002). Seasonal prediction of European spring precipitation from El Niño–Southern Oscillation and local sea-surface temperatures. International Journal of Climatology, 22(1), pp.1-14.

Lloyd-Hughes, B. (2014) The impracticality of a universal drought definition. Theor Appl Climatol 117, 607–611. https://doi.org/10.1007/s00704-013-1025-7

Mann, M. E. et al. (2017) Influence of anthropogenic climate change on planetary wave resonance and extreme weather events. Sci. Rep. 7, 45242.

Martija-Díez, M., Rodríguez-Fonseca, B., & López-Parages, J. (2021). ENSO Influence on Western European Summer and Fall Temperatures, Journal of Climate, 34(19), 8013-8031.

Martija-Díez, M., López-Parages, J., Rodríguez-Fonseca, B. et al. (2022) The stationarity of the ENSO teleconnection in European summer rainfall. Clim Dyn.

McKee, T.B., N.J. Doesken and J. Kleist, (1993): The Relationship of Drought Frequency and Duration to Time Scales. Proceedings of the 8th Conference on Applied Climatology, 17–23 January 1993, Anaheim, CA. Boston, MA, American Meteorological Society.

Kautz, L.-A., Martius, O., Pfahl, S., Pinto, J. G., Ramos, A. M., Sousa, P. M., and Woollings, T.: Atmospheric blocking and weather extremes over the Euro-Atlantic sector – a review, (2022) Weather Clim. Dynam., 3, 305–336, https://doi.org/10.5194/wcd-3-305-2022, 2022.

Kumar, A. (2009), Finite samples and uncertainty estimates for skill measures for seasonal prediction, Mon. Weather Rev., 137, 2622–2631.

Lopez, A. & Haines, S. (2017). Exploring the usability of probabilistic weather forecasts for water resources decision-making in the United Kingdom. Weather, Climate, & Society, 9(4):701-715

Maisey P, Thornton H, Fung F, Harris G, Lowe J, McSweeney C, Mitchell JFB, Murphy J, Rostron J, Sexton D and Yamazaki K. (2018) UKCP18 Factsheet: Weather Types. Met Office Hadley Centre, Exeter

Marsh, T. , Cole, G. and Wilby, R. (2007) Major droughts in England and Wales, 1800-2006. Weather, 62, 87–93.

Masato, G., Hoskins, B.J. and Woollings, T.J. (2012), Wave-breaking characteristics of midlatitude blocking. Q.J.R. Meteorol. Soc., 138: 1285-1296. https://doi.org/10.1002/qj.990

Masato G, Hoskins BJ, Woollings T. (2013) Winter and summer Northern Hemisphere blocking in CMIP5 models. J Clim. 26(18):7044–59.

Matsueda M, Endo H. The robustness of future changes in Northern Hemisphere blocking: a large ensemble projection with multiple sea surface temperature patterns. (2017) Geophys Res Lett.;44:5158–66.

Murphy C, Wilby RL, Matthews TKR, Thorne P, Broderick C, Fealy R, Hall J, Harrigan S, Jones P, McCarthy G, MacDonald N, Noone S, Ryan C. (2020a) Multi-century trends to wetter winters and drier summers in the England and Wales precipitation series explained by observational and sampling bias in early records. Int J Climatol. 2020 Jan;40(1):610-619. doi: 10.1002/joc.6208

Murphy C, Wilby RL, Matthews T, Horvath C, Crampsie A, Ludlow F, Noone S, Brannigan J, Hannaford J, McLeman R, Jobbova E. (2020b) The forgotten drought of 1765-1768: Reconstructing and re-evaluating historical droughts in the British and Irish Isles. Int J Climatol. 2020 Oct;40(12):5329-5351. doi: 10.1002/joc.6521.

Murphy, J.M. and others (2018), UKCP18 Land Projections: Science Report, Met Office, available at https://www.metoffice.gov.uk/pub/data/weather/uk/ukcp18/science-reports/UKCP18-Land-report.pdf

Poli P, Hersbach H, Dee DP, Berrisford P, Simmons AJ, Vitart F et al (2016) ERA-20C: an atmospheric reanalysis of the Twentieth Century. J Clim 29(11):4083–4097.

Pelly JL, Hoskins BJ. A new perspective on blocking. (2003) J Atmos Sci 2003;60(5):743–755.

Osso, A., Sutton, R., Shaffrey, L. and Dong, B. (2018) Observational evidence of European summer weather patterns predictable from spring. Proceedings of the National Academy of Sciences of the United States of America, 115 (1). pp. 59-63. ISSN 0027-8424 https://doi.org/10.1073/pnas.1713146114

Ossó, A., Sutton, R., Shaffrey, L. and Dong, B. (2020) Development, amplification, and decay of Atlantic/European summer weather patterns linked to spring North Atlantic sea surface temperatures. Journal of Climate, 33 (14). pp. 5939-5951. ISSN 1520-0442 doi: https://doi.org/10.1175/jcli-d-19-0613.1

Palmer, W.C., (1965): Meteorological Drought. Research Paper No. 45, US Weather Bureau, Washington, DC.

Parry, S., Wilby, R. L., Prudhomme, C., and Wood, P. J.: A systematic assessment of drought termination in the United Kingdom, (2016), Hydrol. Earth Syst. Sci., 20, 4265–4281, https://doi.org/10.5194/hess-20-4265-2016.

Pope, J.O., Brown, K., Fung, F. et al. Investigation of future climate change over the British Isles using weather patterns. Clim Dyn 58, 2405–2419 (2022). https://doi.org/10.1007/s00382-021-06031-0

Priestley M, Ackerley D, Catto J, Hodges K, McDonald R, Lee R. (2020) An Overview of the Extratropical Storm Tracks in CMIP6 Historical Simulations, Journal of Climate, DOI:10.1175/JCLI-D-19-0928.1

Prodhomme, C., Materia, S., Ardilouze, C. et al. (2022) Seasonal prediction of European summer heatwaves. Clim Dyn 58, 2149–2166. https://doi.org/10.1007/s00382-021-05828-3

Raible, C. C., Lehner, F., González-Rouco, J. F., and Fernández-Donado, L., (2014) Changing correlation structures of the Northern Hemisphere atmospheric circulation from 1000 to 2100 AD, Clim. Past, 10, 537–550, https://doi.org/10.5194/cp-10-537-2014.

Rakovec, O., Samaniego, L., Hari, V., Markonis, Y., Moravec, V., Thober, S., et al. (2022). The 2018–2020 multi-year drought sets a new benchmark in Europe. Earth's Future, 10, e2021EF002394. https://doi.org/10.1029/2021EF002394

Rao, J., Ren, R., Chen, H., Yu, Y., & Zhou, Y. (2018). The stratospheric sudden warming event in February 2018 and its prediction by a climate system model. Journal of

Geophysical Research: Atmospheres, 123, 13,332– 13,345. https://doi.org/10.1029/2018JD028908

Rodda, J.C.; Marsh, T.J.. 2011 The 1975-76 Drought - a contemporary and retrospective review. Wallingford, UK, NERC/Centre for Ecology & Hydrology, 58pp.

Rousi, E., Kornhuber, K., Beobide-Arsuaga, G. et al. Accelerated western European heatwave trends linked to more-persistent double jets over Eurasia. (2022) Nat Commun 13, 3851. https://doi.org/10.1038/s41467-022-31432-y

Scaife A, Arribas A, Blockley E, Brookshaw A, Clark R, Dunstone N, Eade R, Fereday D, Folland C, Gordon M, Hermanson L, Knight JR, Lea DJ, MacLachlan C, Maidens A, Martin M, Peterson AK, Smith D, Vellinga M, Wallace E, Waters J, Williams A. (2014). Skillful long-range prediction of European and North American winters. Geophys. Res. Lett. 41(7): 2514– 2519.

Scaife AA, Comer RE, Dunstone NJ, Knight JR, Smith DM, MacLachlan C, Martin N, Peterson KA, Rowlands D, Carroll EB. (2017) Tropical rainfall, Rossby waves and regional winter climate predictions, Quarterly Journal of the Royal Meteorological Society, volume 143, no. 702, pages 1-11, DOI:10.1002/qj.2910.

Schiemann, R., Athanasiadis, P., Barriopedro, D., Doblas-Reyes, F., Lohmann, K., Roberts, M. J., Sein, D. V., Roberts, C. D., Terray, L., and Vidale, P. L. (2020) Northern Hemisphere blocking simulation in current climate models: evaluating progress from the Climate Model Intercomparison Project Phase 5 to 6 and sensitivity to resolution, Weather Clim. Dynam., 1, 277–292, https://doi.org/10.5194/wcd-1-277-2020.

Schwierz C, Croci-Maspoli M, Davies HC. (2004) Perspicacious indicators of atmospheric blocking. Geophys Res Lett. 31(6)

Schubert, S., Wang, H. & Suarez, M. (2011) Warm season subseasonal variability and climate extremes in the Northern Hemisphere: The role of stationary Rossby waves. J. Clim. 24, 4773–4792.

Shah, Jignesh et al (2022) Increasing footprint of climate warming on flash droughts occurrence in Europe. Environ. Res. Lett. 17 064017

Shaw, T., Baldwin, M., Barnes, E. et al. Storm track processes and the opposing influences of climate change. Nature Geosci 9, 656–664 (2016). https://doi.org/10.1038/ngeo2783

Shepherd, T.G., Boyd, E., Calel, R.A. et al. Storylines: an alternative approach to representing uncertainty in physical aspects of climate change. (2018) Climatic Change 151, 555–571. https://doi.org/10.1007/s10584-018-2317-9

Smith, D.M., Eade, R., Scaife, A.A. et al. (2019) Robust skill of decadal climate predictions. npj Clim Atmos Sci 2, 13. https://doi.org/10.1038/s41612-019-0071-y

Smith, D.M., Eade, R., Andrews, M.B. et al. (2022) Robust but weak winter atmospheric circulation response to future Arctic sea ice loss. Nat Commun 13, 727.

Stagge, J.H., Kingston, D.G., Tallaksen, L.M. et al. (2017) Observed drought indices show increasing divergence across Europe. Sci Rep 7, 14045. https://doi.org/10.1038/s41598-017-14283-2

Spinoni, J. Paulo Barbosa, Alfred De Jager, Niall McCormick, Gustavo Naumann, Jürgen V. Vogt, Diego Magni, Dario Masante, Marco Mazzeschi, (2019) A new global database of meteorological drought events from 1951 to 2016, Journal of Hydrology: Regional Studies, Volume 22, 100593, ISSN 2214 5818, https://doi.org/10.1016/j.ejrh.2019.100593.

Spinoni, J. et al., (2020) Future Global Meteorological Drought Hot Spots: A Study Based on CORDEX Data. Journal of Climate, 33(9), 3635–3661, doi: 10.1175/jcli-d-19-0084.1.

Stringer, N. , Knight, J. and Thornton, H. (2020) Improving meteorological seasonal forecasts for hydrological modeling in European winter. Journal of Applied Meteorology and Climatology, 59(2), 317–332

Sutton, R. and Dong, B. (2012) Atlantic Ocean influence on a shift in European climate in the 1990s. Nature Geoscience, 5. pp. 788-792. ISSN 1752-0908 doi: https://doi.org/10.1038/ngeo1595

Svensson, Cecilia ; Hannaford, Jamie (2019). Oceanic conditions associated with Euro-Atlantic high pressure and UK drought. Environmental Research Communications, 1 (10), 101001. 9, pp. https://doi.org/10.1088/2515-7620/ab42f7

Slivinski, L. C., Compo, G. P., Whitaker, J. S., and others (2019), Towards a more reliable historical reanalysis: Improvements for version 3 of the Twentieth Century Reanalysis system. Q J R Meteorol Soc. ; 145; 2876– 2908. doi:10.1002/qj.3598.

Taylor, C. M., Birch, C. E., Parker, D. J., Dixon, N., Guichard, F., Nikulin, G., and Lister, G. M. S. (2013), Modeling soil moisture-precipitation feedback in the Sahel: Importance of spatial scale versus convective parameterization, Geophys. Res. Lett., 40, 6213– 6218, doi:10.1002/2013GL058511.

Thompson, V. et al. (2017) High risk of unprecedented UK rainfall in the current climate. Nat. Commun. 8, 107.

Tibaldi S, Molteni F. (1990) On the operational predictability of blocking. Tellus A: Dyn Meteorol Ocean. 42(3):343–65.

Touma, D., M. Ashfaq, M.A. Nayak, S.-C. Kao, and N.S. Diffenbaugh, (2015) A multi-model and multi-index evaluation of drought characteristics in the 21st century. Journal of Hydrology, 526, 196–207, doi: 10.1016/j.jhydrol.2014.12.011.

UKCP18 Science Overview Report, (2018), Lowe J. and others, Met Office, available at https://www.metoffice.gov.uk/pub/data/weather/uk/ukcp18/science-reports/UKCP18-Overview-report.pdf

Ulbrich, U., J. G. Pinto, H. Kupfer, G. C. Leckebusch, T. Spangehl, and M. Reyers, (2008) Changing Northern Hemisphere storm tracks in an ensemble of IPCC climate change simulations. J. Climate, 21, 1669–1679.

Varino, F., Arbogast, P., Joly, B. et al. Northern Hemisphere extratropical winter cyclones variability over the 20th century derived from ERA-20C reanalysis. (2019) Clim Dyn 52, 1027–1048. https://doi.org/10.1007/s00382-018-4176-5

Vicente-Serrano SM, Begueria S, Lopez-Moreno JI. (2010). A multi-scalar drought index sensitive to global warming: the standardized precipitation evapotranspiration index – SPEI. J. Clim. 23: 1696 – 1718

Vicente-Serrano, S.M. et al., (2021) Long-term variability and trends in meteorological droughts in Western Europe (1851–2018). International Journal of Climatology, 41(S1), E690–E717, doi: 10.1002/joc.6719.

Wallace, J. M. and Gutzler, D. S., (1981) Teleconnections in the geopotential height field during the Northern Hemisphere winter, Mon. Weather Rev., 109, 782–812

Wang, L., Ting, M. & Kushner, P.J. (2017) A robust empirical seasonal prediction of winter NAO and surface climate. Sci Rep 7, 279. https://doi.org/10.1038/s41598-017-00353-y

Wazneh, H., Gachon, P., Laprise, R. et al. (2021) Atmospheric blocking events in the North Atlantic: trends and links to climate anomalies and teleconnections. Clim Dyn 56, 2199–2221. https://doi.org/10.1007/s00382-020-05583-x

Weisheimer, A., N. Schaller, C. O'Reilly, D. MacLeod and T. Palmer (2017): Atmospheric seasonal forecasts of the 20th Century: multi-decadal variability in predictive skill of the winter North Atlantic Oscillation (NAO) and their potential value for extreme event attribution. Q. J. R. Meteorol. Soc. doi:10.1002/qj.2976.

Wilby, R., (1993). Evidence of ENSO in the synoptic climate of the British Isles since 1880. Weather, 48(8), pp.234-239.

Wilby R, O'Hare G, Barnsley N. (1997). The North Atlantic Oscillation and British Isles climate variability, 1865–1996. Weather 52(9): 266– 276.

Wohland, J., Omrani, N.-E., Witthaut, D., & Keenlyside, N. S. (2019). Inconsistent wind speed trends in current twentieth century reanalyses. Journal of Geophysical Research: Atmospheres, 124, 1931– 1940. https://doi.org/10.1029/2018JD030083

Woollings, T., Hannachi, A. and Hoskins, B. (2010), Variability of the North Atlantic eddy-driven jet stream. Q.J.R. Meteorol. Soc., 136: 856-868. https://doi.org/10.1002/qj.625

Woollings, T., & Blackburn, M. (2012). The North Atlantic Jet Stream under Climate Change and Its Relation to the NAO and EA Patterns, Journal of Climate, 25(3), 886-902.

Woollings, T., Barriopedro, D., Methven, J. et al. (2018) Blocking and its Response to Climate Change. Curr Clim Change Rep 4, 287–300. https://doi.org/10.1007/s40641-018-0108-z

World Meteorological Organization (WMO) and Global Water Partnership (2016): Handbook of Drought Indicators and Indices (M. Svoboda and B.A. Fuchs). Integrated Drought Management Programme (IDMP), Integrated Drought Management Tools and Guidelines Series 2. Geneva.

Wulff CO, Greatbatch RJ, Domeisen DI, Gollan G, Hansen F (2017) Tropical forcing of the summer east Atlantic pattern. Geophys Res Lett 44(21):11166–11173.

Yu, J.-Y., Wang, X., Yang, S., Paek, H. & Chen, M. (2017) The changing El Nino-Southern oscillation and associated climate extremes. Clim. Extremes 226, 3–38.

Zappa, G., Shaffrey, L. C., Hodges, K. I., Sansom, P. G., & Stephenson, D. B. (2013). A Multimodel Assessment of Future Projections of North Atlantic and European Extratropical Cyclones in the CMIP5 Climate Models, Journal of Climate, 26(16), 5846-5862.

Zhang, R., Sutton, R., Danabasoglu, G., Kwon, Y.-O., Marsh, R., Yeager, S. G., et al. (2019). A review of the role of the Atlantic Meridional Overturning Circulation in Atlantic Multidecadal Variability and associated climate impacts. Reviews of Geophysics, 57, 316–375. https://doi.org/10.1029/2019RG000644

B: Past variability in observed hydrological (river flow) drought

Jamie Hannaford, Lucy Barker, Stephen Turner, Simon Parry, Maliko Tanguy, Amulya Chevuturi.

UK Centre for Hydrology & Ecology

Overview

About the review: this report aims to summarise the state of the art on past variability in observed hydrological drought for the UK – in particular, addressing the evidence on whether or not there have been observed changes in drought over time. It aims to provide a summary of what we know and what we do not know about this topic, with the latter providing an idea of key knowledge gaps. The report is organised into a series of topics, and at the end of this report we summarise what we know and do not know under these topics.

What is covered: we address firstly the question of drought definitions and indicators used for drought, before considering the importance of drought propagation (the translation from meteorological to hydrological drought). We then summarise the evidence for past trends and variability, asking the question as to whether droughts have become more severe over time. We look at this over the last 40- 50 years (the period of record of most hydrological observations) before taking a 'long view' over >100 years and more. We then look at the potential drivers of observed changes, addressing firstly climate factors and then human factors (such as changes in abstractions, reservoir influences or land use/land management). We also address the international picture, focusing on studies that include the UK as part of the wider European domain.

What is not covered: we address hydrological (river flow) drought, but not groundwater, which is covered in a separate report. As we are focusing on past variability we primarily focus on observations of hydrological drought, rather than the outputs of simulation models, which are more fully captured in separate reviews. However, we include model-based assessments where appropriate (e.g. where models have been used to extend hydrological records back in time).

Key Findings: There have been many studies looking at hydrological droughts and we have a good toolkit for quantifying them. When there are major drought events, it is often said that droughts are becoming more severe due to anthropogenic warming. While the evidence for human warming is unequivocal, it cannot be said so readily that there have been robust changes in hydrological drought – certainly there is not (yet) strong evidence that droughts have become more severe, despite the occurrence of two major hydrological droughts in the last half-decade. In contrast, there are sound scientific reasons why we should expect changes to hydrological drought in a warming world, and future projections indicate we will. This lack of strong or widespread observational trends does not mean there is no underlying influence of climate change, given the challenges of detecting trends in observational records (mainly because records are relatively short and the 'signal' of climate change can be obscured by significant year-to-year variability). The evidence base for human influences on hydrological drought is weak. While some associations between human influences (water management practices or land use/land cover changes) and drought properties have been found, these are typically not conclusive due to a lack of data on such human interventions.

Recommendations: reconciling past observations and future projections remains a key scientific challenge. A priority towards achieving this will be better understanding of the past drivers of change, both climate-driven and anthropogenic catchment changes, and quantifying their relative roles. This will require integration of field observation and climate and hydrological modelling, as well as further statistical and large-sample hydrological approaches. All these activities critically depend on observational datasets. There have been efforts to improve the observational evidence base (e.g. the UK Benchmark network of near-natural catchments, long-term historical river flow reconstructions). However, major barriers remain (not least information on artificial influences and land use/land cover change) but initiatives are underway to overcome them to provide improved foundations for future studies.

1. Introduction

Throughout much of 2022, the UK experienced one of the most severe droughts in recent decades (Parry et al. 2022). By late summer and early autumn, many catchments had reached low flows not witnessed since the 'benchmark' drought of 1976. This episode followed a major drought in 2018 – 2019 (Turner et al. 2021) and this succession of events has naturally led to claims that such droughts are a manifestation of human-induced global warming. That these droughts have been interspersed with record-breaking floods (e.g. Sefton et al. 2021) suggests a tendency towards more pronounced hydrological extremes in general, which is certainly in keeping with the headline messages of climate change projections (e.g. Watts et al. 2015). The recent droughts have demonstrated the continuing vulnerability of the UK to drought, and underlined the need to understand whether and how drought risk is changing, and how it is likely to evolve in future. Observations of drought and its impacts are central to this objective.

Drought is widely written about as a complex, multi-faceted phenomenon that defies straightforward definition. Since Wilhite and Glantz (1985), drought has commonly been categorised into various types (as noted in the general introduction to these reviews, Allen et al. 2023) often differentiating between meteorological, hydrological, agricultural droughts, alongside various others. This review focuses on hydrological drought. For a thorough introduction to the topic of hydrological drought, the reader is referred to van Loon (2016). More specifically, this review considers only river flow drought, and does not cover groundwater (the subject of a separate review, Bloomfield et al. 2023), lakes, reservoirs and so on. However, for convenience and brevity we use the term hydrological drought throughout.

Why are we interested in river flows? The simple answer is that river flows are one of the primary ways in which climate extremes (like droughts) have an impact on society and the environment, and through which climate change is likely to bring some of its most catastrophic consequences. Adequate river flows (of acceptable quantity and quality) are of fundamental importance to public water supply, abstractions for industry, energy and agriculture, for hydropower generation and for a host of other purposes including navigation and recreation. Moreover, river flows are vital for maintaining healthy aquatic ecosystems, and the many ecosystem services they support. Shortfalls in river flows during hydrological droughts can have impacts for many economic sectors and cause increased competition between them, as well as between human demands and the environment – with subsequent impacts on water, food and energy security in the long-term.

It is also worth highlighting that river flows integrate across a range of processes occurring in a catchment. While many meteorological measurements (notably, raingauges) sample only points in space, river flows represent the combined balance of hydrological fluxes across large areas of the upstream land surface. River flows are, therefore, a key broad-scale indicator of water availability, and long-term measurements of river flow enable us to track hydro-climatic variability on a range of timescales.

Consequently, there is a pressing need to understand river flow droughts – and whether such episodes are becoming more frequent and severe in a warming world. A key aspect of this is in characterising past variability in hydrological drought, to detect emerging trends and provide a baseline against which future changed can be quantified.

In this review, we set out to capture the state-of-the-art in the evidence for past variability in hydrological drought. In keeping with other reviews in this series, we aim to capture what we know and what we do not know about this topic. This framing around variability is important, as hydrological drought is a very broad area – commanding an entire textbook (Tallaksen & van Lanen, 2004) – and hence our focus is specifically on whether, and how, hydrological drought has changed over time. However, as context we also discuss the state of our knowledge of two important underpinning practical topics: hydrological drought definitions and indicators (section 2) and drought processes and propagation (section 3).

We will review the position of our knowledge of how droughts have changed by considering both past trends and variability (Section 4) and drought occurrence (i.e. when did the most severe droughts occur and what were their characteristics) over many decades back to the 19th Century (Section 5). Importantly, we will also consider the mechanisms (or drivers) behind variability in river flow drought. We address climatic drivers (Section 6) and catchment drivers (section 7) – the latter encompassing changes in direct human interventions: abstractions, discharges, reservoir management, land cover changes and so on. These human factors are topical given current debates around 'drought in the Anthropocene' (van Loon et al. 2016), that recognises a central role for humans as agents and aggravators of hydrological drought.

In this review, as we are focusing on past variability we will primarily focus on observations of hydrological drought, rather than the outputs of simulation models, which are more fully captured in separate reviews (Lane et al. 2023; Coxon et al. 2023). We will include model-based assessments where appropriate (e.g. where models have been used to extend hydrological records back in time).

The focus of the review, as commissioned, is on England, but as the majority of the literature considers the UK as a whole, we consider that here so that the review is of wider relevance to readers across the UK. This is especially important considering the large spatial scale of droughts, which are not confined by national borders, and the presence of cross-border flows and water transfers within the UK itself. We also address the international picture, focusing on studies that look at hydrological drought in the UK as part of the wider European domain.

2. Definitions, indicators and methodologies for hydrological drought

Having previously established that we are dealing with hydrological drought, specifically river flow, it is worth briefly revisiting the topic of definitions. There are numerous definitions of the term ‘drought’, but common to all of them is the fact that drought is a relative concept, and must be seen as a departure in precipitation relative to the ‘normal’ conditions for the time of year, for the location in question, sufficient to cause a serious hydrological imbalance (e.g. Tallaksen and Van Lanen, 2004; Mishra and Singh, 2010).

Given this relativistic framing, droughts can be difficult to identify – it can be hard to pin down the onset, duration and termination of a drought event. As a result, it can be difficult to quantify the severity or duration of a drought. To this end, there is a substantial (and growing) literature on the subject of drought indicators and drought indices. A drought indicator can be any variable that is used to characterise drought (e.g. river flow), whereas drought index is normally a numerical representation of drought severity for a given indicator (e.g. a time series accumulated river flows over a given period in cubic metres per second) – see WMO & GWP (2016).

Drought indicators are widely used to assess drought status, or to identify droughts in past hydrological records or future hydrological projections. There is no universal drought indicator (Lloyd-Hughes, 2014 argues that such a notion is impractical by its very nature) and a very wide range have been developed and applied. Lloyd-Hughes (2014) identified over 100 drought indicators in the literature at the time, and numbers have continued to proliferate (see also Bachmair et al. 2016 for a ‘review of reviews’ on drought indicators). Nevertheless, drought indicators are an important part of the toolkit for hydrological drought assessment. We will not review hydrological indicators exhaustively here, but indicators for hydrological drought are described in more detail in reviews of hydrological drought concepts (e.g. Tallaksen & Van Lanen, 2004; Van Loon, 2016) and in a World Meteorological Organization Handbook (WMO & GWP, 2016). Here, we will focus briefly on reviewing indicators and indices of hydrological drought as applied in the UK, highlighting recent trends before identifying key gaps.

In England (as elsewhere in the UK), there is not currently a single fixed definition of drought adopted by all parties. Rather, operational bodies like regulators and water companies have broad classifications of drought status, which are defined according to triggers and thresholds from a wide range of different indicators (meteorological, hydrological, water supply and so on, e.g. (Hannaford et al. 2019; Facer-Childs et al, in review). As such, there is no single over-arching hydrological indicator or index. Commonly, practitioners use simple metrics such as average monthly flows, seasonal flows (often reported as anomalies according to averages, or using ‘return period’ bandings) or low flows (e.g. flow quantiles such as the Q95, or annual minima, typically based on a moving average (e.g. the 7-day minimum flow). Low flows are a key metric of water availability but are not necessarily a ‘drought’ indicator per se, as the lowest flow in each year may not correspond to a drought event (e.g. in a relative wet year), but knowledge of extreme low flows is certainly key information for understanding drought risk.

One of the most widely-used approaches to hydrological drought definition and quantification, internationally, is the threshold level method, proposed by Yevjevich (1967) and widely used (see Tallaksen and Van Lanen, 2004), that allows drought characteristics to be extracted (Fig 1). Such approaches have been used in the UK: e.g. Rudd et al. (2017, 2019), who used the threshold approach to identify past droughts, and also changes in the characteristics in future hydrological projections. One of the appealing properties of extracting droughts in this way is that a frequency distribution can be fitted to the various extracted drought characteristics, allowing the estimation of return periods to drought events according to their duration, intensity (etc.).

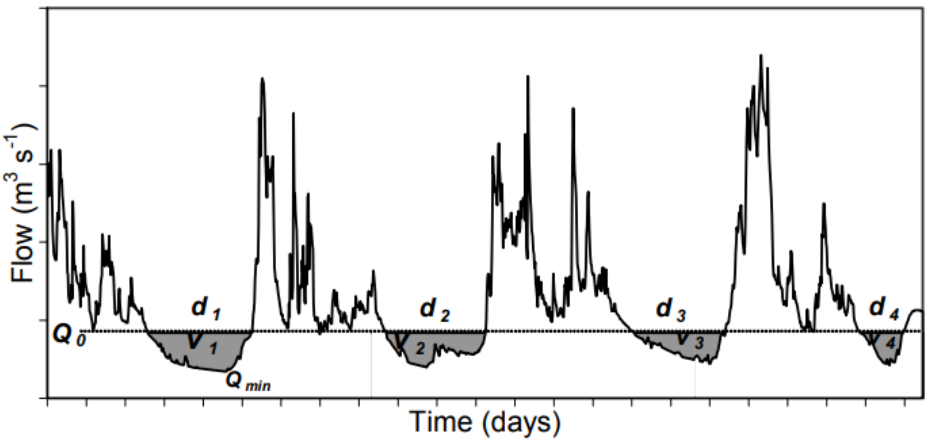

Fig 1: conceptual diagram of the Threshold level approach (Hisdal et al. 2004). The threshold (Q_0, e.g. Q_{90}) is applied based on the long-term record. Drought events are identified when flows drop below this. Event characteristics can be extracted for the periods under the threshold, e.g. duration (d), volume (v) and the intensity or minimum flow (Q_{min})

In recent years, there has been growing adoption of standardised indices. These are based on the Standardized Precipitation Index (SPI, McKee et al. 1993) that is the de facto meteorological drought indicator for drought monitoring and early warning, as recommended by the WMO (Hayes et al. 2011). The SPI has the advantage of enabling comparisons in drought severity in time and space, i.e. between different seasons and locations. The Standardised Streamflow Index (SSI) is directly analogous but applied to river flow time series, and a Standardised Runoff Index (SRI, e.g. Shukla & Wood, 2008) has also been advanced, with the term typically used when applied to runoff from the land surface in gridded hydrological models rather than river flow (although in practice there is some interchangeability).

The SSI was little used in the UK until recently, but following the work of Svensson et al. (2017) and Barker et al. (2016) the index has been more widely used (see also applications by Dobson et al. 2020, West et al. 2021). These indicators are used operationally on the UK Water Resources Portal (Barker et al. 2022; https://eip.ceh.ac.uk/hydrology/water-resources/). An SSI time series can also be used to extract events below a threshold, with the same key characteristics (duration, intensity, etc.) in common with the threshold approach. The SSI has a number of scientific advantages and is seen as a robust approach. However, barriers remain to its widespread adoption, not least around identifying appropriate statistical distributions and interpreting the indicator relative to other, simpler methods like flow percentiles (Svensson et al. 2017; Tijdeman et al. 2020).

While the threshold method and standardised approach have been the mainstays of research in the international literature in recent years, other approaches have been adopted in the UK. A Drought Severity Index (DSI, after Bryant et al. 1994) has been used extensively for meteorological drought assessment, but occasionally for hydrological drought (e.g. Watts et al. 2012). The approach relies on accumulated anomalies below a threshold, and as such shares much with the threshold and SSI approaches. However, it relies on a somewhat arbitrary termination criteria to determine the end of drought events. Other authors have proposed metrics focused specifically on drought termination – Parry et al. (2016a) advocated for the recovery phase of droughts to be featured as characteristics in their own right, advancing a range of metrics to characterise the speed and magnitude of recovery that have since been applied to historical flow records (Parry et al. 2016b).

While there has been an uptick in the number of studies advancing hydrological drought indicators/indices, there is – in common with the wider international picture – as yet little consensus on the 'best' or optimal index for UK hydrological drought. Such consensus may be elusive given the ongoing debate around best ways to index hydrological drought considering the diverse range of impacts of hydrological drought (e.g. Stahl et al. 2016).

There are also debates around which indicators to use for practical applications in monitoring, early warning and forecasting in different sectors in the UK (Hannaford et al. 2019). Fundamentally, there is the question of what such drought indicators really mean in expressing the state or situation of hydrological drought: while drought indices aim at statistical robustness, the practical significance of more rarefied indices can readily be called into question and they can be difficult to interpret (Tijdeman et al. 2020). Previous studies (e.g. Bachmair et al. 2016a) have repeatedly called for (hydrological, among other) drought indices to be 'ground-truthed' against observed evidence of drought impacts. Such studies have been advanced in the UK (Bachmair et al. 2016b, 2017; Parsons et al. 2019) but these efforts are in their infancy given the lack of suitable impact datasets. They have largely focused on meteorological indicators, with limited inclusion of hydrological drought.

Nevertheless, there are significant practical benefits of research focusing on the most appropriate indicators for quantifying hydrological drought. As will become clear in later sections, drought indicators are a cornerstone of our efforts to quantify hydrological drought, how it is changing, and what causes these changes.

3. Hydrological drought processes and propagation

Given even the most loose definitions of drought, all droughts begin with some meteorological anomaly – a deficit in rainfall for the time of year, or due to increased losses due to evaporative demand (or a combination of both), or due to storage of water in ice or snow. From a hydrological perspective, this is only part of the story. The key factor is then drought propagation, i.e. how a meteorological drought propagates through the hydrological cycle to manifest itself in a deficit in river flow. Drought propagation is very well described in textbooks (e.g. Tallaksen and van Lanen, 2004) and reviews such as Van Loon (2016). Propagation implies a delay (lag time) and some degree of smoothing, or lengthening (attenuation) as anomalies travel through the compartments in the hydrological cycle (e.g. Fig 2).

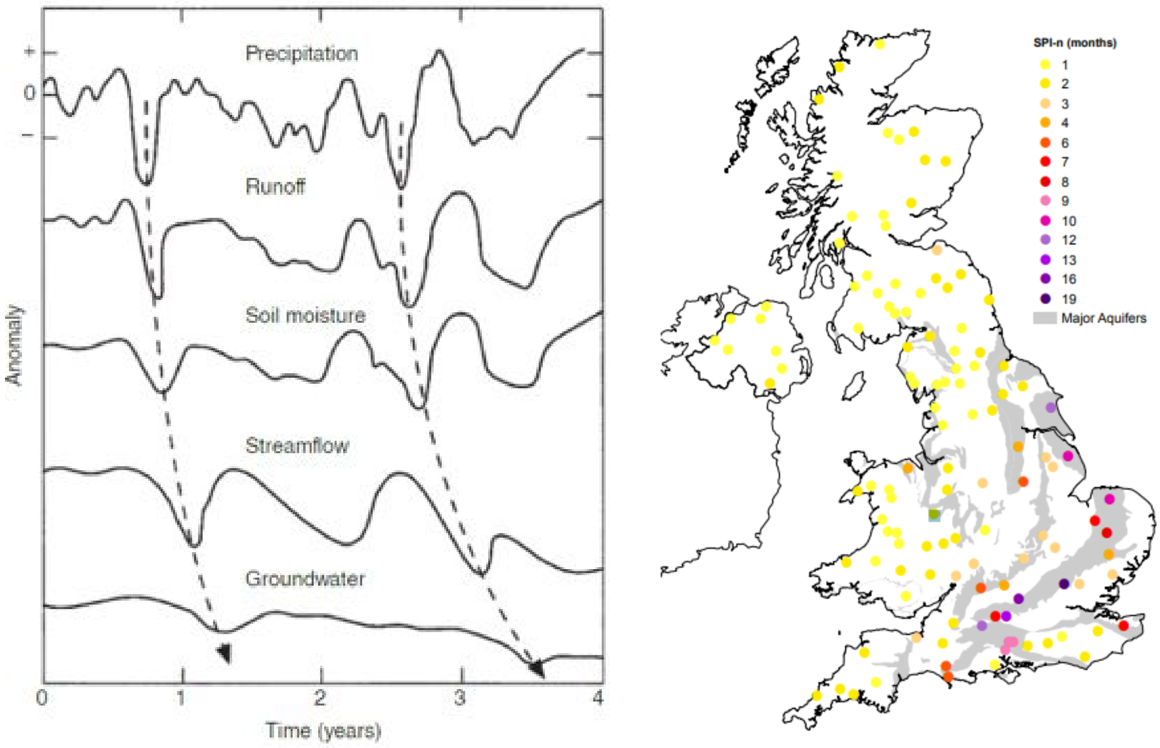

Fig 2: left: A schematic showing how precipitation deficiencies during a hypothetical 4-year period are translated in delayed fashion, over time, through other components of the hydrologic cycle (Chagnon, 1987; reproduced from Tallaksen & Van Lanen, 2004). Right: map showing SPI accumulation period most correlated with SSI-1 for UK Benchmark Catchments (reproduced from Barker et al. 2016). Maps also shows major aquifers.

Drought propagation is of fundamental importance, because the same atmospheric conditions that lead to a precipitation deficit can result in very different river flow responses from different catchments (or within parts of the same catchment), depending on the properties of the area in question – in particular, depending on the nature and configuration of stores in the catchments. It follows, therefore, that meteorological conditions, and attendant indicators (of precipitation, temperature and so on) may be a poor descriptor of hydrological drought responses, and should be treated with caution if the application in question is concerned with river flows. This has important bearings for drought characterisation, monitoring and forecasting, and studies of change in drought (whether past or future) alike.

The question of drought propagation has been very well studied internationally, and has led to typologies of hydrological drought (e.g. van Loon, 2016 and references therein), depending on propagation speed and dynamics, seasonality and the extent to which propagation is controlled by precipitation or temperature anomalies. Such typologies are quite complex and typically designed for European/global applications, and many of the types reflect storage in ice and snow that are (in general) less important in the UK.

Nevertheless, even in the UK, where a great majority of catchments are dominated by ‘classical rainfall deficit’ type droughts (van Loon, 2016), drought propagation processes lead to pronounced regional differences in typical drought characteristics. Generalising broadly, numerous authors differentiate drought vulnerability between the northwest and the southeast – e.g. Jones and Lister, (1998) highlighted how the northwest is vulnerable to single season, ‘within-year’ droughts, while the southeast is typically prone to longer, multi-year events. While there are some inherent differences in typical meteorological drought occurrence across this northwest-southeast gradient, the greatest differences in drought manifest themselves in hydrological terms. Barker et al. (2016) used correlations between standardised indicators to show significant lag-times in hydrological response in the south and east of the UK (see Fig 2, and also Folland et al. 2015). Unsurprisingly, this reflects catchment storages, especially the presence of the Chalk aquifer (see Fig 2). In studies that have sought to identify homogeneous regions defined by drought response, the south east regularly emerges as a distinct cluster(s) represented by slow response times (e.g. Hannaford et al. 2011; Fleig et al. 2011; Svensson & Hannaford, 2019).

It follows that these lag-times in hydrological drought onset also apply in hydrological drought termination (Parry et al. 2016a, b; 2017), with hydrological droughts in many southern and eastern catchments lasting for significant periods after the cessation of meteorological drought – with consequences for drought monitoring, management and communication, as witnessed in practice at the tail end of some recent droughts (e.g. Parry et al. 2013; Turner et al. 2021).

This broad northwest-southeast gradient hides considerable local detail, and significant local variability exists – particularly across the southeast with its heterogeneous hydrogeology. For example, Hannaford et al. (2011) clustered the southeast into two overlapping, non-geographic regions for hydrological drought, with the regions differentiated by catchment responsiveness, according to Base Flow Index (BFI). Chiverton et al. (2016) clustered UK catchments into four clusters using semi-variograms as a way of quantifying catchment responsiveness. There are significant variations between these clusters in hydrological response generally, and in hydrological drought properties in particular (see Barker et al. 2016 who used the same clusters), and a large part of this can be explained by physiographic and hydrogeological properties. However, there is also variability within clusters. Similarly, groundwater studies have highlighted significant heterogeneity in responsiveness and drought characteristics even within the Chalk aquifer (Bloomfield and Marchant, 2013; see also the review of Bloomfield et al. 2023).

Drought propagation also varies through time, even within the same catchment: various studies have shown how the precipitation-flow relationship is non-linear and non-stationary between drought episodes, and even during long multiyear droughts (e.g. Saft et al. 2016, for the millennium drought in Australia), but this has not been investigated in the UK.

Drought propagation normally refers to propagation in time, i.e. a lag time in a given catchment. But we can also recognise drought propagation spatially, i.e. how droughts spread or contract in space as they develop and decay. Droughts are large-scale phenomena, and exhibit complex spatiotemporal evolution. Understanding this is important for monitoring and forecasting drought development (forecasting ‘drought from drought’ by observing neighbouring regions; Hannaford et al. 2011). An understanding of drought spatial coherence is vital underpinning for planning for water transfers as a drought mitigation strategy. There have been several studies of the coherence of meteorological drought (e.g. Rahiz and New, 2012; Tanguy et al. 2021), with the latter highlighting important lessons for spatial drought risk for water transfers – in many historical droughts, typical ‘donor’ regions were under drought conditions simultaneously with the potential recipients. Large-scale coherence between UK regions is only now being assessed quantitatively for hydrological drought (Tanguy et al. 2023). This focused on future projections, but included the recent past as a baseline and again showing significant coherence between potential donor and recipient regions. Tanguy et al. (2023) highlight how such coherence could potentially impact on the feasibility of water transfers as a drought mitigation strategy (although detailed modelling of water supply systems is required to fully quantify such risks; see, e.g. Dobson et al. 2020).

Increasingly, the impacts of humans is recognised as a key factor influencing drought propagation (e.g. Van Loon et al. 2016, 2017). This is discussed further in Section 7, but it is worth noting here that the impact of abstractions, reservoirs and other disturbances can obviously complicate (either aggravating, or countering) the ‘natural’ propagation from meteorological deficit to river flow response (e.g. Tijdeman et al. 2018; Margariti et al. 2019).

In summary, there have been some important advances in understanding drought propagation, and there is much we understand about general relationships between meteorological and hydrological drought. Nevertheless, this is a more complicated topic than it first appears and the literature in this area presents as many questions as it answers. Our knowledge has tended to arise from ‘large-sample’ hydrology studies at the UK scale. Drought propagation has been less well-studied at the catchment scale, although detailed model-based studies have been conducted for some groundwater catchments such as the Pang (e.g. Tallaksen et al. 2009), but there are no studies appraising propagation processes using field observation (to the authors’ knowledge). Given the ‘uniqueness of place’ (e.g. Wagener et al. 2021) constraint in hydrology, further catchment-specific investigation of drought propagation is necessary. Drought propagation is a key science question underpinning the Floods and Droughts Research Infrastructure (FDRI) (Old et al. 2022) so this is likely to be advanced in future.

4. Past variability in hydrological droughts – have droughts become more severe?

In addressing the literature on past changes in drought, it is first important to highlight the very rich information base on which assessments of past changes in hydrological drought is based. England, along with the wider UK, has a very dense hydrometric network in international terms, and is fortunate to have a centralised archive of accessible, quality controlled hydrological data, the National River Flow Archive (NRFA; Dixon et al. 2013; https://nrfa.ac.uk). This resource is the primary basis of most of the studies that have looked at past hydrological variability highlighted in this section.

That said, there are inherent challenges in analysing long-term variability in river flows – as described in Hannaford (2015), Wilby et al. (2017) and Slater et al. (2022). In particular, hydrological records are often impacted by anthropogenic disturbances and constraints of poor data quality. The former is especially important if trying to discern climate-driven changes in river flow. In catchments with strong (or changing) levels of human disturbance, trends and variations may not reflect climate variability. To this end, many countries have declared 'Reference Hydrometric Networks' (RHNs) of near-natural catchments (Burn et al. 2012). The UK was an early leader in this area, with the designation of the UK Benchmark Network (Bradford & Marsh, 2003; updated to UKBN2 by Harrigan et al. 2018). In the following sections, we contrast between some studies that use the Benchmark network and those that apply to a wider range of observations.

A good starting point for any assessment of changing hydrological droughts are the previous Water Report Cards prepared in 2013 – 2015 (Hannaford et al. 2013, 2015; Watts et al. 2013, 2015; see also update by Garner at el. 2017). These reviewed evidence for observed changes in river flow across the UK (including both droughts and floods). These reviews summarised a number of studies of changes in variables such as annual flows, seasonal flows and low flows, with a very mixed picture emerging as far as water resources/drought is concerned – at least compared to high flows/floods where a more consistent picture emerged. Many studies are now quite old and covered data periods ending a decade or more ago. In general, there was limited evidence for any clear trend in annual low flows (e.g. Hannaford et al. 2006, based on data up to 2002). Low flows had typically increased, particularly in the north and west. Seasonal flows showed increases in winter and autumn, decreases in spring, and a very mixed picture in summer (e.g. Hannaford and Buys, 2012, based on data up to 2008). The Report Cards showed that there was little published evidence based around changes in drought (using drought indices like threshold methods/SSI, as opposed to general flow regime indicators).

Since the publication of the Report Cards, there have been few additions to the literature on drought/water resources trends. Harrigan et al. (2018) reviewed and updated the Benchmark Network, and undertook an analysis of seasonal trends and low flows, up to 2016, and found a very similar picture to previous assessments. Both median (Q50) and low (Q95) flows showed increases in northern and western areas but these were rarely significant; decreases were observed across much of England, but these were typically non-significant and there was substantial regional variation. Seasonal flows were consistent with past studies.

While there has been a recent update of flood trends (Hannaford et al. 2021) there has been no published update of low flows or drought trends in parallel. For the purposes of this review, we have undertaken a provisional update of trends in low flows and seasonal flows, comparable with Harrigan et al. (2018) but updated to September 2021 (the latest available data on the NRFA). This was done using the same methodology outlined in Harrigan et al. (2018) and Hannaford et al. (2021). As with Hannaford et al. (2021), we have deliberately compared the UK Benchmark Network (UKBN2) with the wider whole-NRFA network. The time series end in September 2021, as the latest quality controlled NRFA data – this does not feature the 2022 drought, which was among the lowest flows on record (the lowest in some cases) in parts of England and Wales (Parry et al. 2022), but an initial analysis using provisional data from the EA API showed that this made little difference (not shown).

For all the low flow indicators, the same general pattern emerges of increasing flows in northern and western Britain, and a mixed pattern in the English lowlands. However, for the Benchmark network there is a more recognisable tendency towards downward trends. For Q50 and Q70 there are few significant downward trends, but more of the trends in northern Britain are increasing. For Q95, there are some significant downward trends (see Fig 3). Seasonal patterns are similar to previous studies – generally, consistent increases in autumn and winter, and decreases in spring, and a contrast for summer between increases in the north/west and a mixed pattern, but with some significant decreases, in the south. For spring and summer the patterns are similar between the full network and UKBN2 sites, with spring showing decreases across the UK, and summer showing increases in the north/west and decreases in the south/east. For autumn and winter, patterns in the UKBN2 are more mixed, with both increases and decreases in England, although relatively few significant; in Scotland however, all UKBN2 sites show increases.

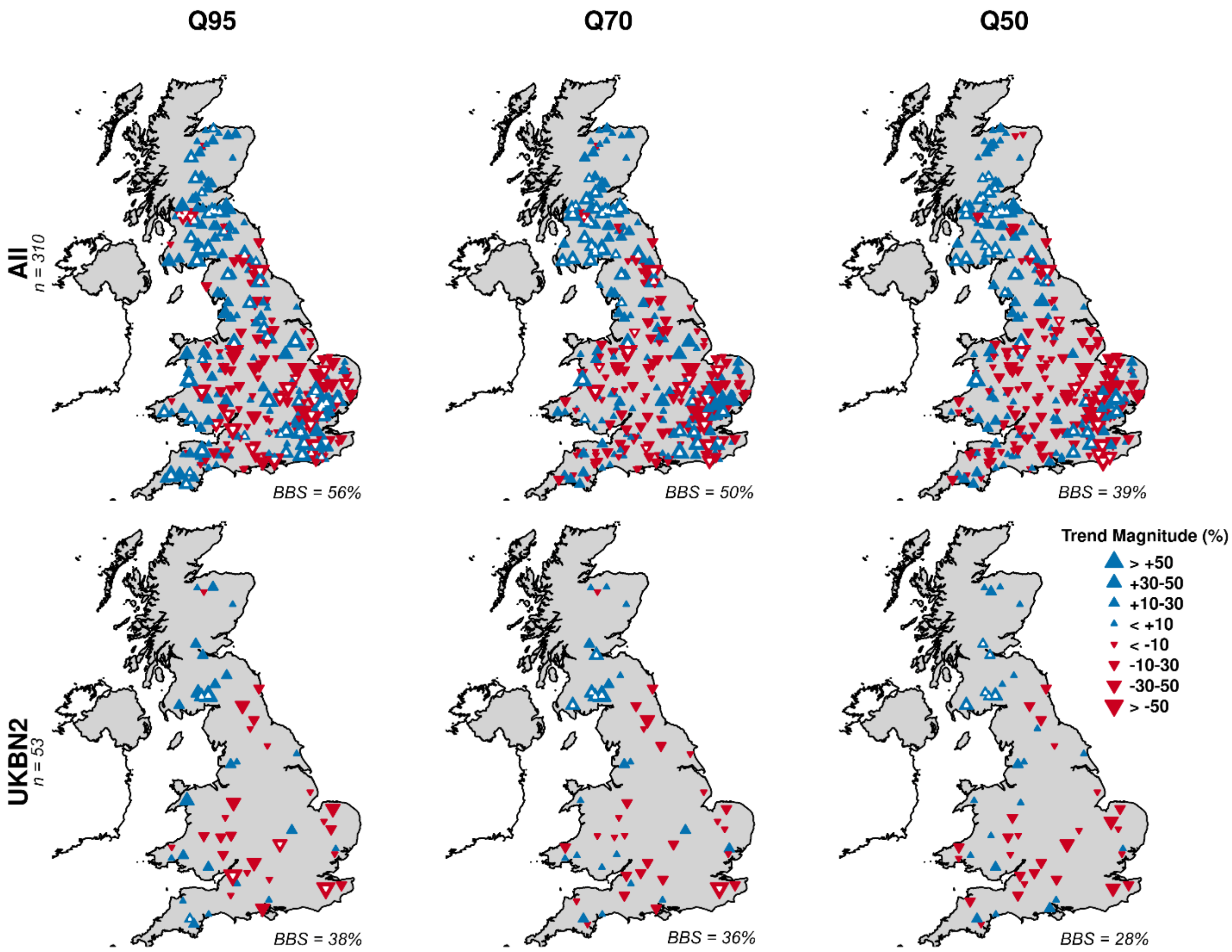

Figure 3: trend analysis of river flow indicators relevant for water resources/drought (Q95, Q70, Q50) for the period 1965 - 2021. Top row = all NRFA catchments with available data (over this period). Bottom row = UK Benchmark Catchments suitable for Low Flow analysis. Trend magnitude is shown according to the key as a percentage change. White colouration of Triangles denotes a significant trend using the Mann-Kendall test (5% level), accounting for serial correlation where present. n.b. These are provisional new results, based on current NRFA data (to end of water year 2020-2021) and using the standard NRFA trend testing approach (for details see Harrigan et al. 2018 and Hannaford et al. 2021).

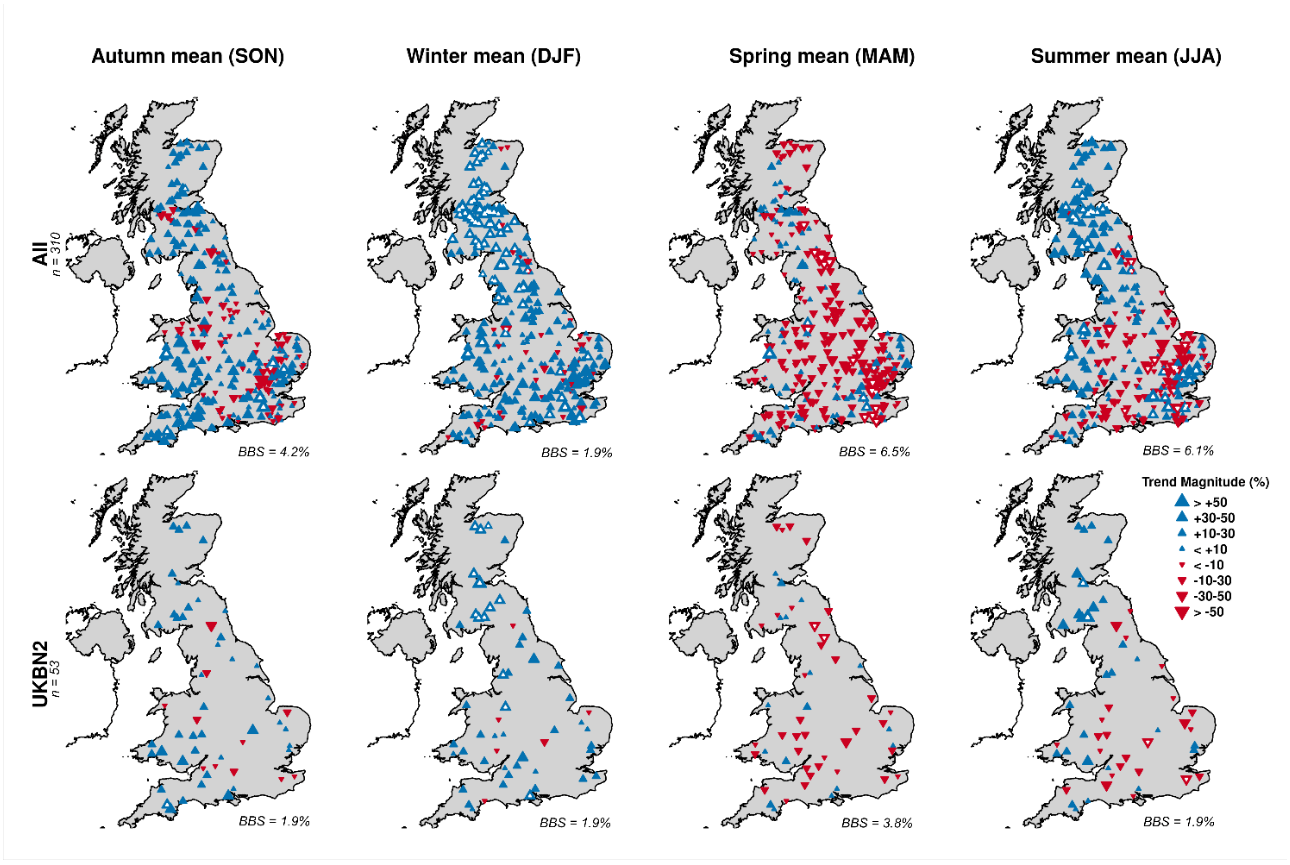

Figure 4: trend analysis of seasonal mean river flows for the period 1965 – 2021 (see figure 3 caption for further explanation)

River flow trends are consistent with observed climate trends, notably significant trends towards wetter winters and, to a lesser extent, autumns, and a pronounced spring drying in the recent past (Kendon et al. 2022). Other studies have also found significant increases in evapotranspiration in spring (Blyth et al. 2019), in addition to spring drying. Summers have, in general, become wetter over the same period as that featured in most river flow studies, but there has been a period of generally wetter summers since c.2007, and drier summers in the 20-30 years before (Kendon et al. 2022). In general, though, river flow trends, like meteorological analyses, shows little compelling evidence (beyond a few catchments with significant downward trends) for any pronounced decreases in summer, nor for low river flows – i.e. the kind of water availability indicators most relevant for drought. This is somewhat at odds with future projections which consistently suggest substantial decreases in summer flow, low flows, and associated increases in drought severity (e.g. summarised in Lane et al. 2023) for the near future.

It should of course be noted that these past studies, and the above new analysis, are of broad indicators of 'drought relevant' seasonal and low flows, rather than analysis of droughts per se, using the kind of indicators highlighted in Section 2. Such studies have not been carried out in detail at the UK scale, yet, although Pena-Angulo et al. (2022) analysed hydrological drought trends between 1962 and 2017 using the SSI, at a European scale, and included 474 UK catchments in their study, embracing a range of both natural and influenced catchments. They found largely negative trends in drought frequency, duration and severity (i.e. towards fewer shorter and less severe droughts) for the UK, albeit also with very mixed patterns. Significant trends towards an amelioration of drought severity were more prevalent in northern and western catchments.

Finally, it is important to underscore that trends are very sensitive to the period of analysis. The new results presented here in Figs 3 and 4, alongside previous studies, typically analyse linear, monotonic trends in a fixed period. Other studies have adopted a ‘multitemporal analysis’ to look at sensitivity of trends to start and end point, and find that varying the start or end by even a few years can radically change the outcomes, with changes in significance and even the direction of change. Hannaford et al. (2021) demonstrate this for flood trends for the UK, but a similar comprehensive analysis of sensitivity to low flow or drought trends is lacking. Wilby (2006) and Hannaford & Buys (2012) showed how varying start years influenced annual, seasonal and low flow trends. In general, trends over the typical ‘observational’ period (post-1960s) are often somewhat different to those seen in longer hydrological records. The increases in summer and low flows seen in many published studies partly reflect the fact that the late 1960s to mid-1970s was notably dry, and the late 1990s – late 2000s was generally much wetter. Murphy et al. (2013) highlight how positive trends are consequently ‘locked in’ by the coverage of typical gauged records in Ireland, and the UK picture is very similar. This underscores the importance of taking a longer view than the typical gauging station record length, as discussed in Section 5.

This sensitivity to study window arises because of strong interannual and interdecadal variability due to a range of large-scale atmospheric/oceanic circulation patterns (see Section 6). This is a possible reason behind the apparent discrepancy between observational trends and future projections for the near-term: Wilby (2006) highlight that it can take very long ‘detection times’ of many decades for a signal of anthropogenic warming to be detectable above the noise of interannual and interdecadal variability. In this context it is unsurprising that ‘detectable’ (i.e. statistically significant) trends have not yet emerged, even if there is an underlying anthropogenic component, and Wilby (2006) argues that trends may be practically significant for water managers way before they become statistically significant.

5. Historical hydrological droughts – a long view

Recent droughts have inevitably invited comparisons with past drought events (e.g. Parry et al. 2022, Turner et al. 2021) and these have shown that 2022 and 2018 droughts rank among some of the most significant hydrological droughts of the last 50-years in terms of low flows. Previous drought events of the 2000s and 1990s were also extensively documented at the time (e.g. 2010 – 2012, Kendon et al. 2013; 2004 – 2006, Marsh et al. 2007) and again, these events were found to be significant in the context of the typical gauged record – that is, from the 1960s/1970s, when the majority of UK gauging stations were installed.

Despite the half-century coverage of many gauging stations, which is impressive in an international context, the ‘instrumental’ record only contains a handful of major drought events. To appraise drought risk more fully, many authors have highlighted the need to examine droughts over much longer timescales. This is important for water resources management, particularly in the context of the deep uncertainty in future climate projections. While the past may not be so readily a guide to the future in a warming world, at the same time observed historical droughts represent an important benchmark of drought risk, given that these events have actually unfolded – they also offer the opportunity to learn from past experiences in drought management. Historical droughts have, therefore, always formed a cornerstone of water resource planning. While recent developments have moved away from a single ‘drought of record, i.e. a worst drought used as a stress test, to considering droughts more severe than the observed envelope (using stochastic methods and other approaches) (Counsell et al. 2023), these methods are ultimately still dependent on past observations – stochastic methods need to be trained on observations. A fuller understanding of historical hydrological droughts is therefore of critical importance to practitioners.

The influential study of Marsh et al (2007) identified major droughts in England and Wales back to 1800. This study highlighted the prevalence of major drought events in the pre-1960 era, and underlined the importance of events such as those of the 1920s, 1930s and the ‘long drought’ period spanning the turn of the 20th century, as well as some droughts in the 1800s which are relatively poorly understood. Marsh et al. 2007 considered drought primarily from a meteorological perspective, given the abundance of long rainfall records – although these authors did gather hydrological evidence, where available, and moreover documented evidence of impact of past drought episodes. From a hydrological viewpoint, such comparisons are challenging given that very few gauging stations captured the droughts of the 1920s – 1940s or earlier.

To fill this gap, there have been several efforts to extend hydrological records through reconstruction, primarily using rainfall-runoff models to estimate past river flows given the long meteorological records available as input. The earliest work of Jones (1984) was updated by Jones et al. (1998) and Jones et al. (2006), and delivered monthly reconstructions (hereafter, CRU reconstructions) back to 1860 for 15 catchments in England and Wales using a simple statistical water balance model driven by long raingauge series. Jones et al. (1998) used the DSI to identify major droughts in these records, and highlighted that in no cases were the contemporary droughts of the 1970s – 1990s the most severe droughts in the longer-term records.

More recently, as part of the ‘Historic Droughts’ project, Smith et al. (2019) delivered a dataset of reconstructed river flows for 303 UK catchments (Historic Droughts reconstructions) using the GR4J hydrological model, driven by a newly-updated high-resolution daily gridded precipitation dataset and Potential Evaporation (PE) reconstructed from gridded temperature (using the approach of Tanguy et al. 2018). Barker et al. (2019) then used these reconstructions to conduct an analysis of historical hydrological droughts and their relative duration and severity using the SSI, for 108 benchmark catchments (Figure 5). In common with previous studies, these authors showed that while recent droughts in the well-gauged era (post-1960) rank highly, there are many historical episodes that are longer or more severe than those of the recent past. A separate reconstruction was conducted for the 'MaRIUS' project by Rudd et al. (2017) using a distributed model, Grid2Grid, also driven by gridded meteorological inputs, and with droughts extracted using a fixed threshold approach. Barker et al. (2019) and Rudd et al. (2017) found, unsurprisingly, good agreement with the droughts identified by Marsh et al (2007). However, these studies highlight important departures, e.g. the importance of droughts in the 1940s that are not well-attested in impact terms due to wartime reporting, and the late 1960s and early 1970s – the impacts of which were eclipsed by the 1976 event. Importantly, both Rudd et al. (2017) and Barker et al. (2019) concluded that there were no obvious, discernible trends in hydrological drought (cf. Fig 5) in these centennial scale reconstructions. However, no formal trend tests were carried out.

These studies also provide a consistent and robust way of understanding the relative importance of historical hydrological droughts, which is important for water resources planning. Barker et al. (2019) quantitatively appraised the relative severity of past hydrological droughts (in terms of both duration, three severity characteristics and for two different SSI averaging periods, SSI3 and SSI12), and how this varied around the country. These authors found the 'hydrological drought of record' varies around the country, and that this depends also on the 'type' of drought, i.e. whether looking at short duration (SSI3) or long duration (SSI12) droughts, and which property is being considered (duration or intensity) (Fig 6).

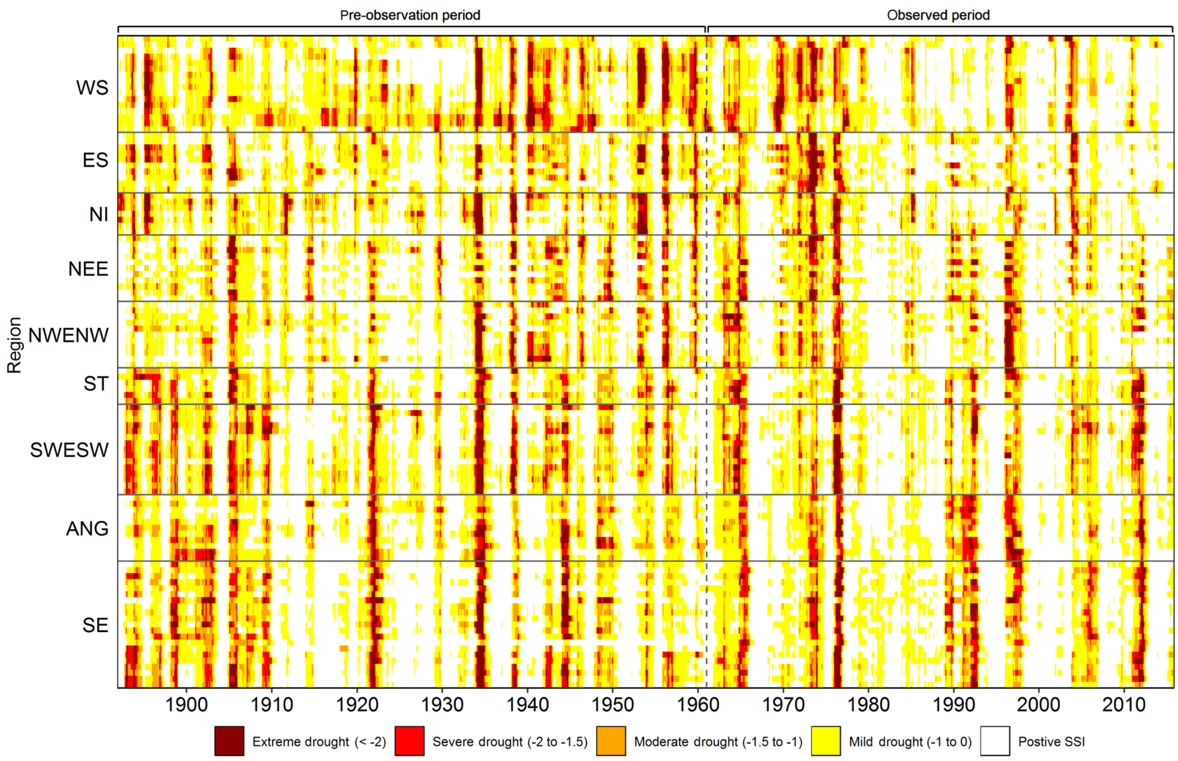

Fig 5 – Heat map of SSI-12 for LFBN catchments from 1891 to 2015 (catchments arranged roughly from north to south on the y axis, with one row per catchment and hydro-climatic regions marked for clarity) with colours according to SSI12 category in key. 'Observed period' highlights typical maximum record coverage of most

gauging stations. 'pre-observation' the period with most added value from the reconstructions. Reproduced from Barker et al. 2019

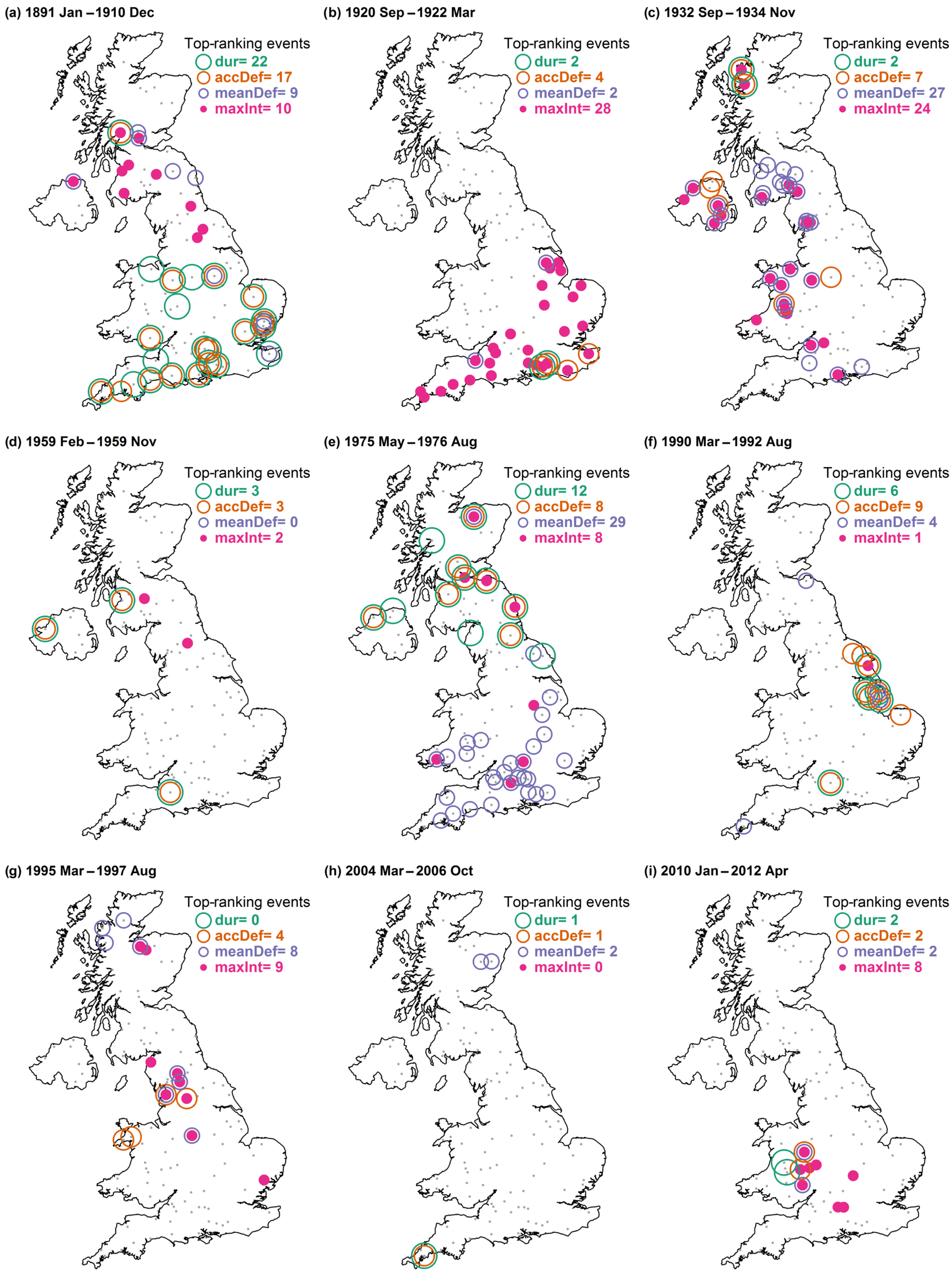

Fig 6: Location and number of LFBN catchments, where the top-ranking SSI-12 event corresponds to major known drought events, for duration (dur), accumulated deficit (accDef), mean deficit (meanDef) and maximum intensity (maxInt). Each of

the nine maps represents one of the major drought events of Marsh et al. 2007. Each point on the maps represents the location of the 108 LFBN catchments. Points are coloured pink where the particular event was ranked most severe according to each event characteristic. Reproduced from Barker et al. (2019)

Following on from this theme of identifying ‘droughts of record’ for water resources planning, several other noteworthy studies have reconstructed hydrological droughts on a regional basis, and then fed these into water supply system models, e.g. for East Anglia (Spraggs et al. 2015) and the Midlands (Lennard et al. 2015). Interestingly, in both cases it was found that an extended reconstruction of droughts into the 19th Century made little difference to water supply yields – that is, the additional 19th century droughts did not test water supply systems more than those in the available long rainfall records (generally, back to the 1920s). However, these conclusions are regional and system-specific, so further research would be needed to see if the Historic Droughts/MaRIUS reconstructed hydrological droughts make a significant difference in other parts of the country.

Finally, although reconstructions have enriched our understanding of past hydrological droughts, they still extend only to 1865 (CRU reconstructions) or 1890 (Historic Droughts and MaRIUS reconstructions). Reconstructions have not been attempted, yet, for earlier periods. This is an opportunity, given recent advances in extending meteorological datasets further into the 19th century (Hawkins et al. 2022). Comparable river flow reconstructions in Ireland have been developed from 1766 (O’ Connor et al. 2022), suggesting credible hydrological drought reconstructions can be made over these very long time horizons.

6. Drivers of change in hydrological drought – climate factors

Trends and past variations in river flows such as those described in section 5 and 6 can be driven by either climate or non-climate (catchment) factors. Some effort to isolate the climate-driven signal has been made through the identification of Benchmark catchments. However, having established a ‘control’ network for detecting climate-driven changes, the question remains of what mechanism is behind the observed river flow change. Most pertinently, the question is whether observed changes are attributable to anthropogenic warming, or due to variability in the wide range of natural, internally forced modes of ocean-atmosphere variability. More realistically given the extent to which these factors are intrinsically linked, the answer is some combination of both, and the question is whether the relative roles can be disentangled and quantified. This is not an abstract question, as the time evolution of future trends will depend on the balance between ‘thermodynamic’ anthropogenic warming, which is unidirectional to all intents and purposes, and circulation-driven changes which may moderate or counter such trends. Such questions are arguably among the most challenging questions in hydrology (several of the ‘twenty three unsolved problems in hydrology’ of Bloschl et al. (2019) spring from this, while Wagener et al.

(2021) argue that credibly discerning climate impacts on hydrology is one of six key knowledge gaps in our understanding of the terrestrial water cycle of Great Britain).

In this section, we briefly review the literature on the hydroclimatology of UK droughts, i.e. on climate-river flows associations, to understand what climate factors have been linked to variations in UK river flows. Knowledge of this topic is central to the climate detection and attribution debate, and yet is also of practical importance for the development of monitoring and seasonal forecasting systems.

Arguably the most extensively studied climate-hydrological associations are those connections, or teleconnections, between river flows and larger-scale, lower frequency modes of variability – atmospheric circulation indices such as the North Atlantic Oscillation (NAO). The NAO is the leading mode of variability in the euro-Atlantic sector, and as such is an obvious candidate for linking with river flows. The NAO, through its strong control of the location of the storm track and thus moisture delivery to the British Isles, has long been shown to strongly influence UK rainfall, especially in the winter months, and it follows that river flow patterns can also be linked to NAO variability. There is a large literature on this topic which we will not cover in detail here. But this literature is consistent in showing very similar patterns, namely a strong positive association between the NAO Index (NAOI) and river flow in the winter months, especially in northern and western areas. However, relationships are complex, especially in non-winter months, and especially in the lowlands of southern and eastern England, where the effect of the NAO is modest and, again, strongly catchment-controlled (e.g. Laize and Hannah, 2010; West et al. 2021). The NAO is not the only relevant pattern, and other studies have shown a prominent role of other modes of variability (notably the East Atlantic pattern and the Scandinavia pattern, e.g. Hannaford et al. 2011; West et al. 2022). West et al. (2022) linked NAO and EA patterns to the SPI and SSI, and highlighted the interaction of these modes of variability, throughout the year, and note how their relative role varies around the country as well as seasonally – as well as the role of propagation from SPI to SSI. While the NAO dominates in winter in the north and west, it has far less explanatory power in the south and east in summer, when the EA plays a key role in modulating the NAO influence.

The upshot of the strong control of the NAO, EA and other modes of variability is that the time evolution of river flows, and drought indicators to an extent, can be seen to be controlled by the variability and interplay of these patterns. A prominent role for the NAO has been claimed for explaining trends towards wetter winters (and higher river flows) in northern and western UK (e.g. Hannaford et al. 2015, and references therein) over the 1960s – late 1990s especially when the NAO was primarily positive. However, since then the NAOI has been more variable yet trends towards higher winter flows have been unabated. The picture is a very complex one, and recent studies have shown strong non-stationarity in the relationship between the NAO and UK rainfall and river flows (as well as groundwater levels) over long timescales (e.g. Rust et al. 2022).

While the dipole-based NAO, EA, SCA and synoptic scale drivers can explain some variability of hydrological drought occurrence, there is arguably even greater benefit from zooming out still further to consider the role of larger-scale, slowly varying ocean-atmosphere drivers - notably (quasi-) cyclical patterns of sea-surface temperature variations such as El Nino-Southern Oscillation (ENSO) or the Atlantic Multidecadal Oscillation (AMO) that themselves influence the state of the NAO. Such patterns have a reasonable degree of predictability, so uncovering robust links between them and river flow could have profound implications for efforts to forecast and project water availability. Folland et al. (2015) reviewed the state of knowledge of such links at the time, and demonstrated links between ENSO, and a range of other predictors, and UK (specifically, lowland England) rainfall – most notably with La Nina events (links which have been long established; see references therein). They also showed the impacts of La Nina on river flows and groundwater, including drought indicators like the SPI/SSI for the Thames region. While links between La Nina events and English lowlands winter half-year droughts were uncovered, such relationships are weak and highly non-linear.

More recently, Svensson and Hannaford (2019) also took a global scale approach to explore links between UK regional rainfall and river flows on the one hand, and SST patterns in both the Atlantic and Pacific oceans. These authors confirmed an impact of Pacific Ocean variability (the Pacific Decadal Oscillation, strongly linked to ENSO), but found it was highly modulated by the state of the North Atlantic (Figure 7). Such relationships were present not just for the winter, but in summer months, previously considered much less promising for forecasting, and yet of the most importance for drought management. The implication is that to understand UK river flow variability, and hydrological drought, it is necessary to look well beyond WTs or even dipole-like circulation indices, and zoom out to take a global view of atmosphere-ocean dynamics.

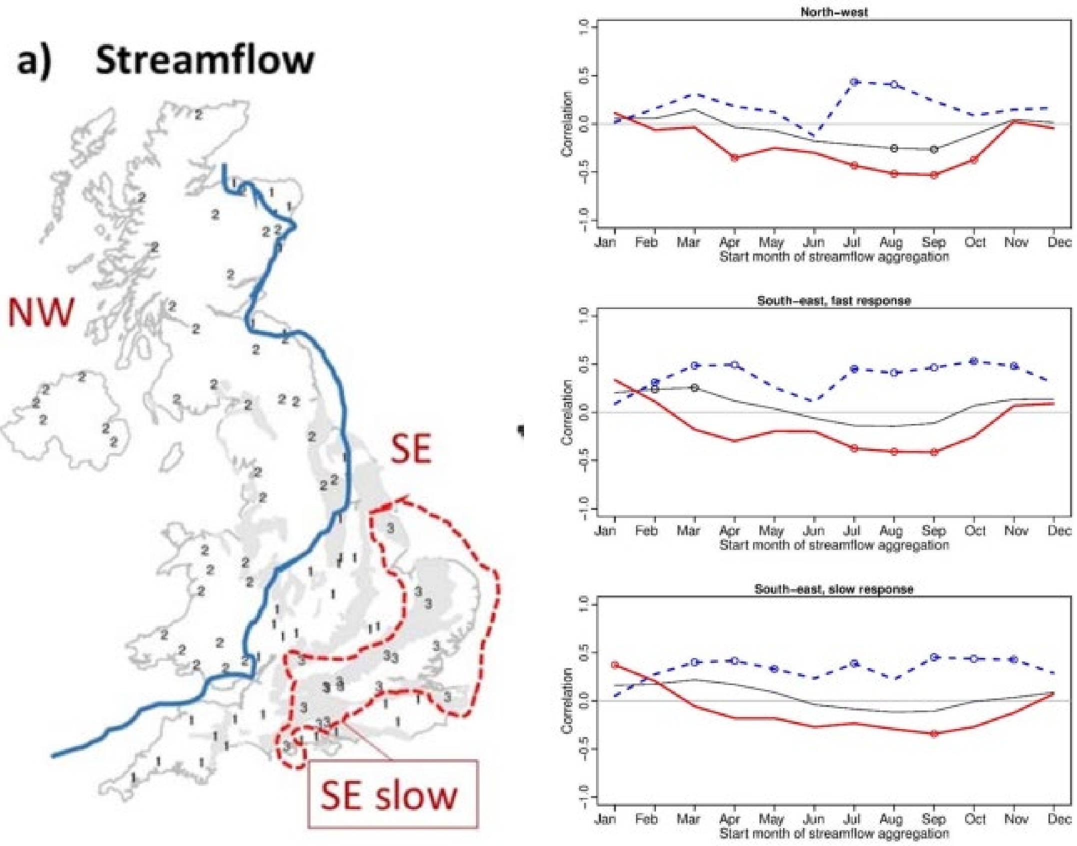

Figure 7: Relationships between ocean-atmosphere conditions and UK river flows (modified from Svensson & Hannaford, 2019). Left: three homogeneous regions emerging from cluster analysis. Right: correlations between three-month

aggregations of PDO and streamflow, with streamflow lagged three months after the PDO, for three different UK regions, 1961–2016. The correlations are stratified on AMO phase: AMO < −0.05 (dashed blue line); AMO −0.05 (thick solid red line); all AMO regardless of phase (thin solid black line). Significant correlations are circled.

In general, then, there have been some advances in explaining the drivers of hydrological drought through relating various climate/ocean indices to river flow indicators. Few studies, however, have linked to drought indicators specifically. In addition, while such relationships have been used to explain observed river flow variability and trends, most have been what may be termed ‘soft attribution’ through associations and correlation. There have been few ‘hard attribution’ studies (Merz et al. 2012), that is, studies that have demonstrated conclusively a causal chain between climate variations and trends in river flow (‘proof of consistency’, Merz et al. 2012) and also ruled out other factors (proof of inconsistency) – e.g. catchment changes, as discussed in section 7

A second aspect of attribution is separating any signal of anthropogenic warming from internally-forced variations such as ENSO, AMO and so on, discussed above. Formal climate detection and attribution studies have been undertaken for UK flood events (e.g. for the 2013-2014 floods; Schaller et al. 2016, Kay et al. 2016) but not for meteorological drought (e.g. rainfall anomalies) for any UK drought event – although human fingerprints on heat extremes are well established, including for the 2022 UK annual temperature (Christidis et al. 2023). Given the role of evaporation losses in exacerbating hydrological droughts, it is likely that anthropogenic warming has played a role but attribution has not been attempted. While formal detection and attribution studies have been undertaken for meteorological drought globally (e.g. Chiang et al. 2021), they have not been applied for hydrological indicators. A majority are also event-based rather than attributing long-term trends. Gudmundsson et al. (2021) claimed global trends in mean and low river flows could be attributed to climate warming, but such studies need replicating on a finer scale before proven for the UK.

7. Drivers of change in hydrological drought – human factors

As shown in Section 6, there is a substantial and growing literature on the links between climate drivers and hydrological drought, motivated by the need to understand the factors controlling large-scale water availability. In many UK catchments (in common with many other domains, globally), however, river flows patterns sometimes deviate markedly from climate variability due to pervasive artificial influences on river flow regimes. While RHNs enable climate signals to be discerned, many RHN sites are small, headwater catchments in the uplands, and are often some distance away from major population centres. Arguably the most important locations are those in the heavily populated, intensively managed lower reaches, where understanding climate and human controls on hydrological drought is much more challenging.

Hence, while RHNs seek to filter out artificial influences as a ‘control’, these influences are worthy of study in and of themselves. This has been the spirit of the International Association of Hydrological Sciences (IAHS) ‘Panta Rhei’ decade (https://iahs.info/Commissions--W-Groups/Working-Groups/Panta-Rhei) that has sought to understand and quantify human influences on flow regimes, and that has spawned a ‘drought in the Anthropocene’ initiative (van Loon et al. 2016, 2016b). Internationally, many studies have attempted to quantify the impact of influences such as reservoirs, abstractions, discharges and other regulation on flow regimes and, thence, on drought characteristics (for example, see the overview of van Loon et al. 2022). Such surveys highlight the many challenges in discerning the impact of any particular human influence because multiple impacts occur in parallel, are difficult to disentangle and may offset or compensate for one another.

Nevertheless, in spite of these challenges, these are not just academic debates, but topics of huge societal import: in the UK, there is a long-standing, and sometimes polarised and contentious, debate on the role of abstractions on hydrological drought and low flows, especially for Chalk streams, that has attained particular prominence in recent years (e.g. CaBa, 2021).

Despite this growing interest, in both academia and the public eye, there have been relatively few UK studies in the scientific literature that have conclusively linked artificial influences (or, commonly, a change in artificial influences) with hydrological drought responses. Partly, this reflects the challenges of obtaining suitable datasets of artificial influences. In the absence of directly available datasets of influences, researchers have resorted to indirect techniques. Tijdeman et al. (2018) took a large-sample approach to compare the drought regimes of catchments classified according to the presence/absence of certain influences, using the NRFA’s Factors Affecting Runoff or FAR codes. While the study suggested that deviations in drought regime (i.e. expected response to precipitation) could be linked to influences (notably, extended drought durations linked to the presence of groundwater abstractions in Chalk catchments; Fig 8), in practice the method was primarily a screening approach, and no quantitative proof could be offered in the absence of data on impacts.

Bloomfield et al. (2021) took a large sample approach, using the CAMELS-GB dataset, which does incorporate some limited artificial influences data within, to develop statistical models to assess the impact of abstractions, discharges and reservoir operations on baseflow in 429 catchments. Inclusion of such water management interventions improved the statistical models in some cases – especially for groundwater abstractions, suggesting a detectable impact, in common with Tijdeman et al. 2018. These authors note that more detailed information on water management than is currently available in CAMELS-GB would be needed to fully constrain the specific effects of individual water management interventions on BFI. Recently, Salwey et al. (2023) took a large sample approach to detect reservoir impacts on river flows using hydrological signatures, including low flow metrics. They compared signatures from 111 Benchmark catchments with 186 catchments modified by reservoirs. They found that reservoirs create deficits in the water balance and alter seasonal flow patterns, while low flow variability was dampened by reservoir operations. This approach of comparing signatures between Benchmark and impacted datasets enabled identification of thresholds above which the reservoir 'signal' could be isolated from wider hydroclimate variability, and holds promise for discerning the effect of other human impacts.

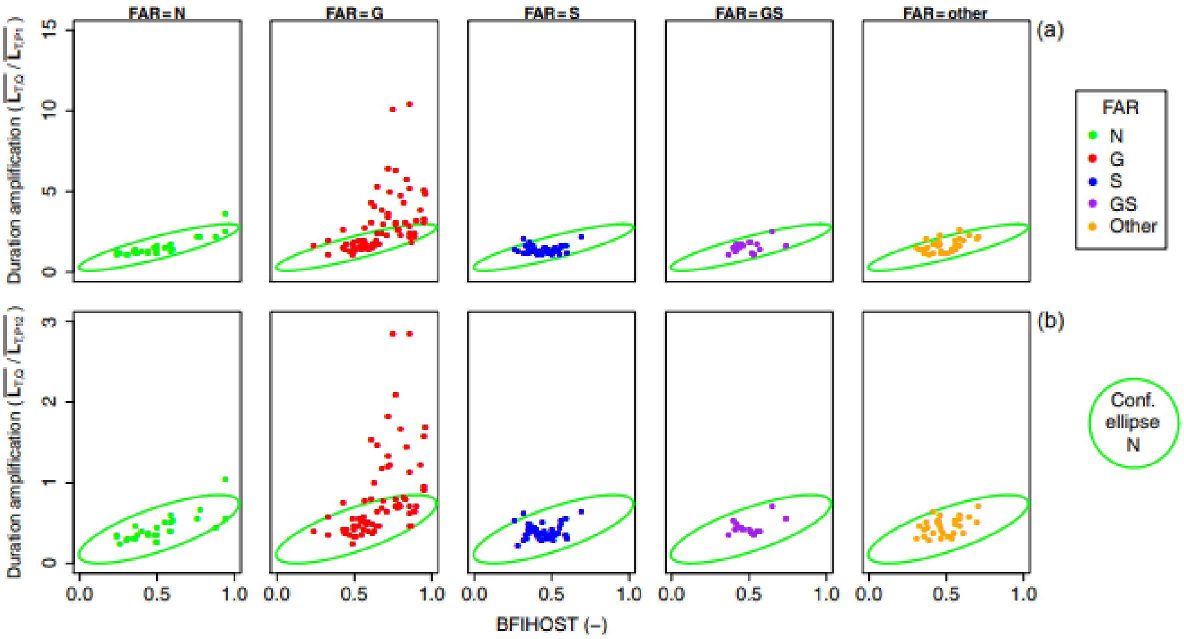

Figure 8: amplification of average monthly streamflow drought duration over average monthly precipitation drought duration (top: 1 month precipitation; bottom: 12 month precipitation) versus BFIHOST for catchments labelled with different 'Factors Affecting Runoff' codes (colours). Ellipse reflects the 95 % confidence ellipse for catchments with near-natural flow records (FAR = N). FAR = G (groundwater abstraction) shows many catchments have longer droughts than expected based on precipitation

Other studies have adopted paired catchment analyses – e.g. van Loon et al. (2019) who compared droughts in two hydrologically-similar catchments in eastern England, with one catchment impacted by a water transfer scheme. While differences can be observed in drought characteristics, once again there is limited information to prove the effect conclusively ('weak attribution' in the parlance of Merz et al. 2012).

It follows that there are few studies that show a change or trend in UK river flows, or relevant drought indicators, that can be attributed to artificial influences, beyond the observation of Tijdeman et al. (2018) of a tendency towards increased drought anomalies over time in many catchments affected by groundwater abstraction. The reverse may also apply when abstraction decreases. Clayton et al. (2008) noted an increase in river flows since the cessation of a major groundwater abstraction in the river Ver, as part of an alleviation of low flow (ALF) scheme, but again noted this could not be confidently attributed to that cause alone. Similarly, Tijdeman et al. (2018) show a similar example for the Darent, a river with an ALF scheme (Figure 9), although also conclude that such relationships need further work to fully elucidate.

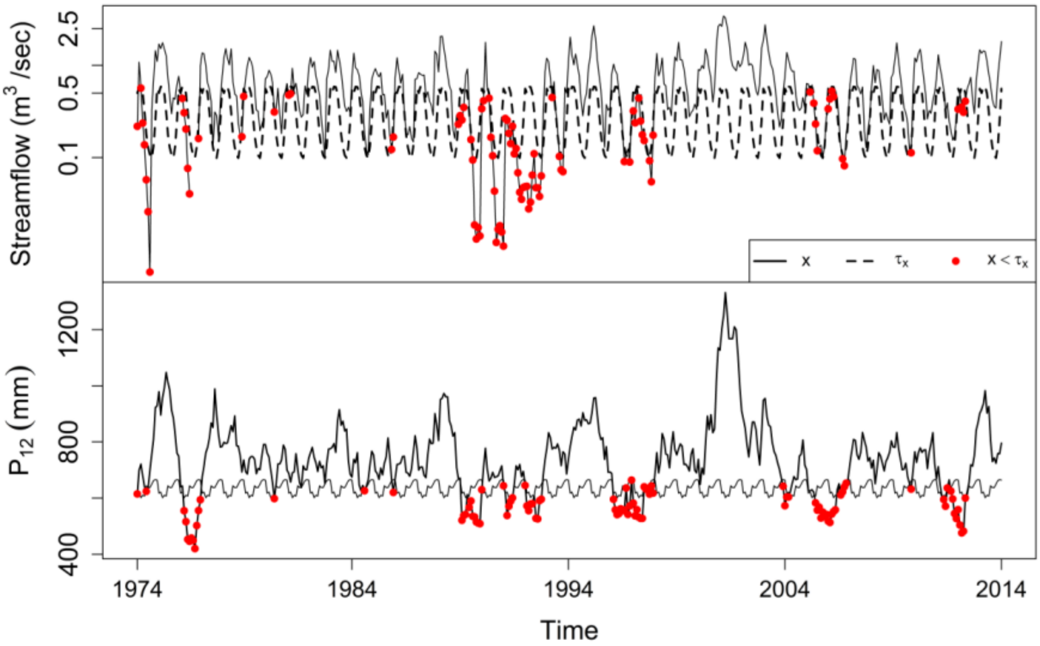

Figure 9: example of possible change in hydrological drought as a result of changing abstractions. Top: hydrological droughts, indexed by the threshold method. Bottom: droughts from threshold method from precipitation (averaged over 12 months). Discrepancies in the droughts of the 1990s likely reflect the impact of abstractions, which has subsequently diminished due to low flow alleviation measures. Tijdeman et al. (2018) – supplementary information.

While the literature on artificial influence impacts on drought is relatively sparse, the situation is even more acute for the influences of land use or land cover (LULC) change, despite this being a long-standing topic in UK (and global) hydrology. This is certainly the case for low flow and drought indicators, that have arguably been neglected in comparison to floods, for which there have been many studies. Nevertheless, reviews and meta-analyses show that there is very limited consensus on the extent to which flood indicators are conclusively influenced by rural land management (e.g. O'Connell et al. 2007), afforestation (Stratford et al. 2017) or Natural Flood Management (Dadson et al. 2017). For water resources or drought indicators, there have been no major efforts to synthesise the literature in a comparable way.

At the catchment scale LULC have been very comprehensively investigated, for isolated catchments – with the most notable example being the paired catchment studies at Plynlimon, mid-Wales (see the review of Robinson & Rodda, 2013). The Plynlimon experiment did not investigate drought responses per se, but showed the impact of afforestation on catchment evaporative losses and, hence, river regimes, including low flows. While there has been growing interest in quantifying the effect of afforestation on flood regimes, as a potential mitigation strategy, there have been few studies looking at drought or low flows at the larger scale. Recently, Buechel et al. (2022) used (land cover) scenarios of potential afforestation applied to a land-surface model (JULES) to quantify the effect of afforestation in twelve diverse (and generally large) UK catchments. Surprisingly, given vigorous debates on the topic, these authors found little impact on flooding, but much larger impacts at median and low flows. It must be noted this was a scenario-based ('what if' scenarios) rather than observational study.

Urbanisation is a major potential impact on streamflow regimes, but again the focus has largely been on investigating the effect of urbanisation on flood frequency (e.g. Prosdocimi et al. 2017). Few studies have investigated wider streamflow regimes more generally. However, in an interesting development, a recent study by Han et al. (2022) investigated non-stationarity in observed river flow regimes in twelve urbanising catchments (with appropriate datasets of changing urban cover) and found that the strongest signals to emerge were for low flows rather than high flows.

In summary, the impact of human interventions on hydrological drought is, perhaps, a hot topic that invites hot takes, but one which rests on a very limited evidence base. One major limitation has been the availability of impact datasets. There have been significant advances in developing datasets of impacts of abstractions and discharges for England, based on the Environment Agency's data holdings – notably CAMELS-GB (Coxon et al. 2020) and the gridded dataset of Rameshwaran et al. (2021). Barriers remain to access, but these are important community assets and further studies will no doubt emerge using them. Even with such data, conclusive attribution will be challenging. The impacts of humans on river flows is a key strand of FDRI (Wagener et al. 2021) – an intensive campaign of integrated field observation and modelling will be needed to make advances in this area.

8. Hydrological drought – fitting UK into a European Perspective

The purpose of this section is to contextualise the UK-focused perspective of the preceding sections. The UK has slotted into a very wide range of continental-scale studies covering all these topics – from assessments of drought typology/propagation, through long-term trends/historical variability into assessments of climate and, to a lesser extent, human drivers of hydrological drought.

The first major European-wide drought assessments were conducted in the 1990s in the EU ARIDE Project, which set out to gather a 'European Water Archive' of river flow data. Several studies span off from this, including studies of the large-scale drivers of hydrological drought, in particular the NAO (Shorthouse and Arnell, 1999) and detailed studies of drought propagation, including for the 1976 and 1990 drought events (Zaidman et al. 2002). A first trends study looking at low flows and drought was published by Hisdal et al. (2004). This was consistent with most domestic studies published soon after, in finding little evidence for trends in drought in the UK or at the wider European scale.

A further EU framework initiative, WATCH, led to numerous innovations. A drought catalogue, based on a Regional Deficiency Index, was delivered by Hannaford et al. (2011), in part also funded by an Environment Agency project looking at spatial coherence of drought across Europe. The drought catalogues showed how hydrological variability in the five UK regions mirrored many of the regions of northern France, the Low Countries and southern Scandinavia. This study investigated large scale patterns (NAO, EA and SCA) driving hydrological drought in Europe, showing how relationships within the UK fit into much wider dipole-like patterns of variations in Europe. Since then studies looking at river flow/climate indices (NAO, EA, SCA) on European scale have also further demonstrated significant non-stationarities in these relationships with river flow (Lorenzo Lacruz et al., 2022).

WATCH also delivered a near-natural RHN across Europe, which the Benchmark network fed into (Stahl et al. 2010). This was used to study trends in river flows – albeit not drought indices, per se, but monthly flows and low flows. This showed how patterns in UK were comparable with northern Europe: generally, positive trends in river flow were found for the UK winter months, in common with most of northern Europe. Negative trends were found in summer months for the UK, while very mixed patterns were found for low flows and low flow timing. Interestingly, these trends appeared somewhat different to the outcomes of national scale studies at the time, notably Hannaford & Buys, 2012 who found little evidence of any summer decreases. Such differences point to how weak trends are, and how influenced they can be by minor changes in study periods.

Much more recently, an expanded dataset (in terms of spatial extent, but not an RHN) has looked at trends in drought characteristics, using the SSI (Peña-Angulo et al. 2022). This study found a generally negative trend towards drought in the UK (as noted in section 4 above), i.e. towards less severe droughts, and showed (as with Stahl et al. 2010, 2012) that this was part of a trend across Europe towards increasing water availability in northern latitudes, compared to decreasing water availability, and worsening hydrological drought, in the south (Fig 9).

The dataset used by Peña-Angulo et al. 2022 is notable for including many impacted catchments rather than just RHNs. Vicente-Serrano et al. (2021) used the same dataset to look at trends in annual runoff, also finding, unsurprisingly, a north-south contrast. These authors tried to explain the causes of these variations, through looking a deviations from expectation (i.e. where meteorological and hydrological trends diverge). Interestingly, these authors attributed the decreases in southern Europe to changes in irrigation and natural re-vegetation following land abandonment, as well as meteorological drying. Trends in the north were found to be more driven by meteorology, although some isolated deviations were noted, but without any attribution. This is another example of a ‘large-sample’ attribution study with limited recourse to appropriate impact datasets, but points to further screening methods that can identify likely hotspots of human impacts.

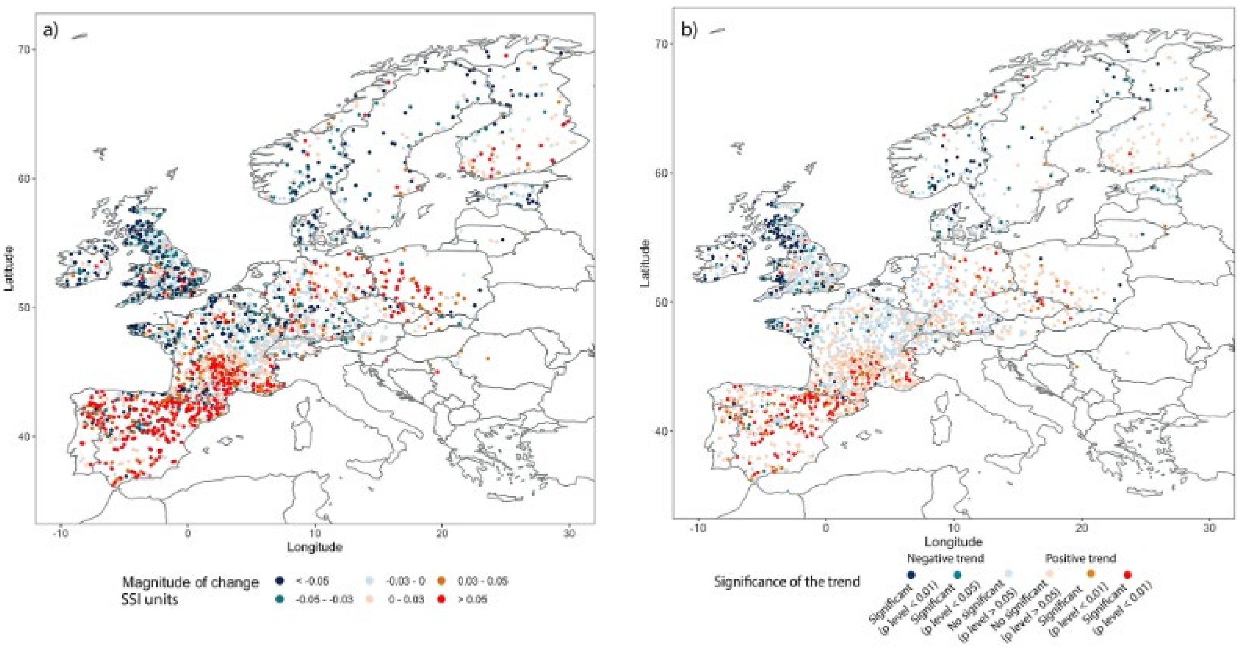

Figure 9: Trends in the severity of drought events from 1962 to 2017. (a) Spatial distribution of the magnitude of change in Standardized Streamflow Index (SSI) and (b) the corresponding significance of trends (at p < 0.05, p < 0.01) over the same period. Each circle represents one gauging station. From Pena-Angulo et al. 2022

9. Summary: what we know and don't know

Definitions and Indicators

- There have been significant advances in the development of indicators and indices to characterise hydrological drought, including threshold method and standardised indices – we know how to apply these indicators in a UK setting and generally when and where they work for what purposes
- That said, there are still things do not know about the practical application of hydrological indices, e.g. the best statistical distributions for the SSI across all seasons or environments, the best accumulation periods to represent impacts on various sectors.
- There are still significant divides between hydrological drought definition and indicators in research, and that applied in drought management and communication.
- There is no 'standard' hydrological indicator, and there is not likely to be convergence on such a thing - arguably there should not be. However, we do not really know which indicators are best suited to hydrological drought applications because there have been only limited efforts to link such indicators through to 'on the ground' impacts. Linking hydrological indicators to impact datasets should be a key priority, and there are opportunities to exploit new and emerging datasets to facilitate this.

Drought Processes and Propagation

- We have a good general understanding of how the propagation of meteorological to hydrological deficits varies around the UK and some understanding of how this is influenced by catchment characteristics. This has helped inform the development of improved monitoring and early warning systems, and has helped underpin our understanding of how catchment vs climate drivers influence changes over time (see section 6 and 7).
- This understanding mostly emerges from 'large-sample' type studies that have correlated time series of precipitation and river flow indicators for a range of catchments, and then further correlated variations between them with different catchment properties.
- We do not fully understand the variability of catchment responsiveness and what catchment processes drive this, nor do we understand how this responsiveness varies between drought events (between moderate and severe droughts, say) or how propagation can be influenced by human influences on river flow.
- We have some understanding of spatial coherence in meteorological droughts, but this has not yet been fully translated to hydrological drought, which is necessary given the complexities of drought propagation. Spatial coherence is important underpinning for water management, especially in the context of water transfers.
- Improved understanding of propagation processes will require concerted field observation as much as larger-sample statistical hydrology and modelling – there will be a major opportunity to explore this in upcoming investments such as FDRI (for which drought propagation is a key science priority, Wagener et al. (2021)).

Observed Trends

- We have a good understanding of how UK streamflow has evolved from the late 1960s to present, for a very wide range of indicators, emerging from a very rich body of research.
- We know that winter and autumn river flows have increased, and spring flows have decreased. Summer shows more mixed trends. We know that these trends are in line with meteorological trends.
- We know that low flows have generally increased in the north and west of the UK, and decreased in the south and east – but relatively few trends are statistically significant.
- However, there have been only limited studies that have really sought to characterise the basic question of 'have droughts become more severe' – the answer to this is still, really, we do not yet know (at least, in comparison with floods which have been studied more comprehensively). There is limited evidence on which to conclude droughts have become demonstrably more severe, and more work is needed to answer this fully.
- There is little compelling evidence that summer flows, nor low flows, have decreased significantly. This is somewhat at odds with climate projections for the near future. We cannot yet fully explain this (apparent) disparity. We know future projections are uncertain. We know past trends are weak and it takes time for signals to emerge from interannual variability. But we have not yet fully tried

to reconcile past trends with future changes comprehensively, and this should be a major priority to enable suitable guidance for water resources planners that bridges the gap between relatively short-term observations and very uncertain model outcomes.

Longer-term historical perspectives

- We know that recent drought events are not necessarily the most severe on record if we take a longer historical view of hydrological drought. Long reconstructed records show that the hydrological 'drought of record' is often one that predates gauging station records.
- We have a good understanding, now, of the likely most severe hydrological droughts for given drought types (short or multiannual) and how this varies around the country
- We do not yet have a full understanding of the significance of these historical droughts for water supply planning around the UK
- We do not have a good understanding of hydrological drought evolution and dynamics before 1890 – improved meteorological driving datasets provide a good opportunity to advance this in future.

Climate drivers

- We understand the main drivers of variability in drivers of river flow variability: we know that that the NAO and EA are important drivers of streamflow variability, and know where and when they are most influential, and how this varies by catchment type (in broad terms)
- However we do not fully understand how these patterns interact to cause hydrological drought events, and how this interaction varies and how it is modulated by drought propagation
- We understand that atmosphere-ocean drivers in both the Pacific (ENSO, PDO) and Atlantic (AMO and Atlantic SSTs) can influence hydrological droughts, and increasingly that there is a need to look globally and at the interaction between these phenomena.
- We do not know, however, the relative importance of all these influences and how they affect river flows in different parts of the UK at different times of year.
- Despite these advances in hydroclimatology of droughts, we do not know to what extent the time evolution of river flows (and it follows, hydrological droughts) on interannual or interdecadal timescales can be confidently linked to variations in these climatic drivers
- As such, we only have a very weak understanding of the extent to which recent variations in river flow reflect internal forcing, anthropogenic warming, or (as is most likely) a combination of both. Quantifying these components will be an important topic for research given the relative combination of these factors will influence future trajectories of UK drought.

Anthropogenic drivers

- We have some understanding of likely impacts of certain types of influence (e.g. heavy abstraction) on hydrological drought – but this understanding emerges indirectly from either large-sample hydrological studies, or paired catchment studies limited to a few sites.

- We have some understanding of the likely impacts of land use changes and interventions, through conceptual understanding as well a few examples of long-term observational studies and various scenario modelling initiatives.
- The literature on LULC impacts on drought and low flows is very sparse compared to flooding – yet some recent studies hint that the impact of LULC on low flows may be more detectable than for high flows.
- We do not know how such human interventions influence river flow regimes, in general, and hydrological drought in particular, across a wide range of UK catchment settings. Not only is this a major constraint in itself, for the majority of heavily influenced catchments, we also do not know the relative contribution of climatic variability versus human interventions in driving variability in hydrological drought, and how this varies regionally and nationally, or how the balance has changed over time.
- Very few studies have conclusively proved a link between changes in anthropogenic drivers (whether water management, or LULC) and river flow responses relevant to hydrological drought, largely due to a lack of data on the relevant interventions. However, there have been relatively few efforts to study the impact of such interventions on hydrological drought. New, emerging artificial influences datasets present a good opportunity to improve on this, if accessibility of such datasets can be secured.

International Perspective

- Trends in river flow, including low flows and drought, for the UK are broadly consistent with neighbouring areas and are part of a wider European picture of a (general) dipole between increasing flows in northern Europe and decreasing flows in the south, albeit with much finer scale and seasonal variation. The climate drivers of UK droughts are also consistent with the wider European picture.
- There are indications that human drivers of low river flows/drought play an important role over and above climate in southern Europe, due to irrigation and revegetation, but no confidence in discerning these drivers further north

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References

Bachmair, S., Stahl, K., Collins, K., Hannaford, J., Acreman, M., Svoboda, M., Knutson, C., Smith, K. H., Wall, N., Fuchs, B., Crossman, N. D., & Overton, I. C. (2016a). Drought indicators revisited: the need for a wider consideration of environment and society. WIREs Water, 3(4), 516–536. https://doi.org/https://doi.org/10.1002/wat2.1154

Bachmair, S., Svensson, C., Hannaford, J., Barker, L. J., & Stahl, K. (2016b). A quantitative analysis to objectively appraise drought indicators and model drought impacts. Hydrology and Earth System Sciences. https://doi.org/10.5194/hess-20-2589-2016

Bachmair, S., Svensson, C., Prosdocimi, I., Hannaford, J., & Stahl, K. (2017). Developing drought impact functions for drought risk management. Natural Hazards and Earth System Sciences, 17(11), 1947–1960. https://doi.org/10.5194/nhess-17-1947-2017

Barker, L. J., Hannaford, J., Chiverton, A., & Svensson, C. (2016). From meteorological to hydrological drought using standardised indicators. Hydrology and Earth System Sciences. https://doi.org/10.5194/hess-20-2483-2016

Barker, L. J., Hannaford, J., Parry, S., Smith, K. A., Tanguy, M., & Prudhomme, C. (2019). Historic hydrological droughts 1891-2015: Systematic characterisation for a diverse set of catchments across the UK. Hydrology and Earth System Sciences, 23(11), 4583–4602. https://doi.org/10.5194/hess-23-4583-2019

Barker, L. J., Fry, M., Hannaford, J., Nash, G., Tanguy, M., & Swain, O. (2022). Dynamic High Resolution Hydrological Status Monitoring in Real-Time: The UK Water Resources Portal. Frontiers in Environmental Science, 10. https://www.frontiersin.org/articles/10.3389/fenvs.2022.752201

Beven, K. J. (2000). Uniqueness of place and process representations in hydrological modelling. Hydrology and Earth System Sciences, 4(2), 203–213. https://doi.org/10.5194/hess-4-203-2000

B Bloomfield, J. P., & Marchant, B. P. (2013). Analysis of groundwater drought building on the standardised precipitation index approach. Hydrology and Earth System Sciences, 17(12), 4769–4787. https://doi.org/10.5194/hess-17-4769-2013

Bloomfield, J. P., Gong, M., Marchant, B. P., Coxon, G., & Addor, N. (2021). How is Baseflow Index (BFI) impacted by water resource management practices? Hydrology and Earth System Sciences, 25(10), 5355–5379. https://doi.org/10.5194/hess-25-5355-2021

Bloomfield et al. 2023. Review Essay, Groundwater

Blyth, E. M., Martínez-de la Torre, A., & Robinson, E. L. (2019). Trends in evapotranspiration and its drivers in Great Britain: 1961 to 2015. Progress in Physical Geography: Earth and Environment, 43(5), 666–693. https://doi.org/10.1177/0309133319841891

Bradford, R. B., & Marsh, T. J. (2003). Defining a network of benchmark catchments for the UK. Proceedings of the Institution of Civil Engineers - Water and Maritime Engineering, 156(2), 109–116. https://doi.org/10.1680/wame.2003.156.2.109

Bryant, S. J., Arnell, N. W., & Law, F. M. (1994). The 1988–92 Drought in its Historical Perspective. Water and Environment Journal, 8(1), 39–51. https://doi.org/https://doi.org/10.1111/j.1747-6593.1994.tb01091.x

Burn, D. H., Hannaford, J., Hodgkins, G. A., Whitfield, P. H., Thorne, R., & Marsh, T. (2012). Reference hydrologic networks II. Using reference hydrologic networks to assess climate-driven changes in streamflow. Hydrological Sciences Journal, 57(8), 1580–1593. https://doi.org/10.1080/02626667.2012.728705

Buechel, M., Slater, L., & Dadson, S. (2022). Hydrological impact of widespread afforestation in Great Britain using a large ensemble of modelled scenarios. Communications Earth & Environment, 3(1), 6. https://doi.org/10.1038/s43247-021-00334-0

CaBa (Catchment-based approach) partnership (2021). Chalk Stream restoration strategy, 2021. https://catchmentbasedapproach.org/learn/chalk-stream-strategy-3/

Chiang, F., Mazdiyasni, O., & AghaKouchak, A. (2021). Evidence of anthropogenic impacts on global drought frequency, duration, and intensity. Nature Communications, 12(1), 2754. https://doi.org/10.1038/s41467-021-22314-w

Chiverton, A., Hannaford, J., Holman, I., Corstanje, R., Prudhomme, C., Bloomfield, J., & Hess, T. M. (2015). Which catchment characteristics control the temporal dependence structure of daily river flows? Hydrological Processes, 29(6), 1353–1369. https://doi.org/10.1002/hyp.10252

Christidis, N. Stott, P, McCarthy, M. An attribution study of the UK mean temperature in 2022. https://www.metoffice.gov.uk/binaries/content/assets/metofficegovuk/pdf/research/climate-science/attribution/uk-2022-attribution.pdf

Clayton, H. J., Morris, S. E., McIntyre, N. R., & Greaves, M. (2008). The hydrological impact of low-flow alleviation measures. Proceedings of the Institution of Civil Engineers - Water Management, 161(4), 171–180. https://doi.org/10.1680/wama.2008.161.4.171

Coxon, G., Addor, N., Bloomfield, J. P., Freer, J., Fry, M., Hannaford, J., Howden, N. J. K., Lane, R., Lewis, M., Robinson, E. L., Wagener, T., & Woods, R. (2020). CAMELS-GB: hydrometeorological time series and landscape attributes for 671 catchments in Great Britain. Earth Syst. Sci. Data, 12(4), 2459–2483. https://doi.org/10.5194/essd-12-2459-2020

Dadson, S. J., Hall, J. W., Murgatroyd, A., Acreman, M., Bates, P., Beven, K., Heathwaite, L., Holden, J., Holman, I. P., Lane, S. N., O'Connell, E., Penning-Rowsell, E., Reynard, N., Sear, D., Thorne, C., & Wilby, R. (2017). A restatement of the natural science evidence concerning catchment-based 'natural' flood management in the UK. Proceedings of the

Royal Society A: Mathematical, Physical and Engineering Sciences, 473(2199), 20160706. https://doi.org/10.1098/rspa.2016.0706

Dobson, B., Coxon, G., Freer, J., Gavin, H., Mortazavi-Naeini, M., Hall, J.W. 2020. The Spatial Dynamics of Droughts and Water Scarcity in England and Wales. Water Resources Research, 56, e2020WR02718, https://doi.org/10.1029/2020WR027187

Dixon, H., Hannaford, J., & Fry, M. J. (2013). The effective management of national hydrometric data: experiences from the United Kingdom. Hydrological Sciences Journal, 58(7), 1383–1399. https://doi.org/10.1080/02626667.2013.787486

Facer-Childs et al. Reviewing Approaches to Drought status And Risk – the RADAR project Literature Review. Unpublished report to Defra, 120pp

Fleig, A. K., Tallaksen, L. M., Hisdal, H., & Hannah, D. M. (2011). Regional hydrological drought in north-western Europe: linking a new Regional Drought Area Index with weather types. Hydrological Processes, 25(7), 1163–1179. https://doi.org/https://doi.org/10.1002/hyp.7644

Folland, C. K., Hannaford, J., Bloomfield, J. P., Kendon, M., Svensson, C., Marchant, B. P., Prior, J., & Wallace, E. (2015). Multi-annual droughts in the English Lowlands: a review of their characteristics and climate drivers in the winter half-year. Hydrol. Earth Syst. Sci., 19(5), 2353–2375. https://doi.org/10.5194/hess-19-2353-2015

Garner, G., Hannah, D. Watts, G. 2017. Climate change and water in the UK: Recent scientific evidence for past and future change. Progress in Physical Geography, 41, 2. https://doi.org/10.1177/0309133316679082

Gudmundsson, L., Boulange, J., Do, H. X., Gosling, S. N., Grillakis, M. G., Koutroulis, A. G., Leonard, M., Liu, J., Müller Schmied, H., Papadimitriou, L., Pokhrel, Y., Seneviratne, S. I., Satoh, Y., Thiery, W., Westra, S., Zhang, X., & Zhao, F. (2021). Globally observed trends in mean and extreme river flow attributed to climate change. Science, 371(6534), 1159–1162. https://doi.org/10.1126/science.aba3996

Han, S., Slater, L., Wilby, R. L., & Faulkner, D. (2022). Contribution of urbanisation to non-stationary river flow in the UK. Journal of Hydrology, 613, 128417. https://doi.org/10.1016/j.jhydrol.2022.128417

Hannaford, J., & Marsh, T. (2006). An assessment of trends in UK runoff and low flows using a network of undisturbed catchments. International Journal of Climatology, 26(9), 1237–1253. https://doi.org/10.1002/joc.1303

Hannaford, J., Collins, K., Haines, S., & Barker, L. J. (2019). Enhancing drought monitoring and early warning for the United Kingdom through stakeholder coinquiries. Weather, Climate, and Society, 11(1), 49–63. https://doi.org/10.1175/WCAS-D-18-0042.1

Hannaford, J., Lloyd-Hughes, B., Keef, C., Parry, S., & Prudhomme, C. (2011). Examining the large-scale spatial coherence of European drought using regional indicators of precipitation and streamflow deficit. Hydrological Processes, 25(7), 1146–1162. https://doi.org/10.1002/hyp.7725

Hannaford, J. (2015). Climate-driven changes in UK river flows: A review of the evidence. Progress in Physical Geography: Earth and Environment, 39(1), 29–48. https://doi.org/10.1177/0309133314536755

Hannaford, J., Mastrantonas, N., Vesuviano, G., & Turner, S. (2021). An updated national-scale assessment of trends in UK peak river flow data: how robust are observed increases in flooding? Hydrology Research, 52(3), 699–718. https://doi.org/10.2166/nh.2021.156

Hannaford, J., & Buys, G. (2012). Trends in seasonal river flow regimes in the UK. Journal of Hydrology, 475, 158–174. https://doi.org/10.1016/j.jhydrol.2012.09.044

Harrigan, S., Hannaford, J., Muchan, K., & Marsh, T. J. (2017). Designation and trend analysis of the updated UK Benchmark Network of river flow stations: the UKBN2 dataset. Hydrology Research, 49(2), 552–567. https://doi.org/10.2166/nh.2017.058

Hidsal, H., Tallaksen, L. M., Clausen, B., Peters, E., & Gustard, A. (2004). Chapter 5: Hydrological Drought Characteristics. In L. M. Tallaksen & H. A. J. van Lanen (Eds.), Hydrological drought: processes and estimation methods for streamflow and groundwater. Elsevier.

Hawkins, E., Burt, S., McCarthy, M., Murphy, C., Ross, C., Baldock, M., Brazier, J., Hersee, G., Huntley, J., Meats, R., O'Grady, J., Scrimgeour, I., & Silk, T. (2022). Millions of historical monthly rainfall observations taken in the UK and Ireland rescued by citizen scientists. Geoscience Data Journal. https://doi.org/10.1002/gdj3.157

Hayes, M., Svoboda, M., Wall, N., & Widhalm, M. (2011). The Lincoln declaration on drought indices: universal meteorological drought index recommended. Bulletin of the American Meteorological Society, 92(4), 485–488. https://doi.org/10.1175/2010BAMS3103.1

Jones, P. D. (1984). Riverflow reconstruction from precipitation data. Journal of Climatology, 4(2), 171–186. https://doi.org/10.1002/joc.3370040206

Jones, P. D., & Lister, D. H. (1998). Riverflow reconstructions for 15 catchments over England and Wales and an assessment of hydrologic drought since 1865. International Journal of Climatology, 18(9), 999–1013. https://doi.org/10.1002/(SICI)1097-0088(199807)18:9<999::AID-JOC300>3.0.CO;2-8

Jones, P. D., Lister, D. H., Wilby, R. L., & Kostopoulou, E. (2006). Extended riverflow reconstructions for England and Wales, 1865–2002. International Journal of Climatology, 26(2), 219–231. https://doi.org/https://doi.org/10.1002/joc.1252

Kendon, M., Marsh, T., & Parry, S. (2013). The 2010–2012 drought in England and Wales. Weather, 68(4), 88–95. https://doi.org/https://doi.org/10.1002/wea.2101

Kendon, M., McCarthy, M., Jevrejeva, S., Matthews, A., Sparks, T., & Garforth, J. (2022). State of the UK Climate 2021. International Journal of Climatology, 41(S2), 1–76. https://doi.org/10.1002/joc.7787

Laizé, C. L. R., & Hannah, D. M. (2010). Modification of climate–river flow associations by basin properties. Journal of Hydrology, 389(1), 186–204. https://doi.org/10.1016/j.jhydrol.2010.05.048

Lane et al. 2023. Essay on future drought

Lennard, A. T., Macdonald, N., Clark, S., & Hooke, J. M. (2015). The application of a drought reconstruction in water resource management. Hydrology Research, 47(3), 646–659. https://doi.org/10.2166/nh.2015.090

Lloyd-Hughes, B. (2014). The impracticality of a universal drought definition. Theoretical and Applied Climatology, 117(3), 607–611. https://doi.org/10.1007/s00704-013-1025-7

Lorenzo-Lacruz, J., Morán-Tejeda, E., Vicente-Serrano, S. M., Hannaford, J., García, C., Peña-Angulo, D., & Murphy, C. (2022). Streamflow frequency changes across western Europe and interactions with North Atlantic atmospheric circulation patterns. Global and Planetary Change, 212, 103797. https://doi.org/10.1016/j.gloplacha.2022.103797

Margariti, J., Rangecroft, S., Parry, S., Wendt, D. E., & van Loon, A. F. (2019). Anthropogenic activities alter drought termination. Elementa: Science of the Anthropocene, 7. https://doi.org/10.1525/elementa.365

Marsh, T., Cole, G., & Wilby, R. (2007). Major droughts in England and Wales, 1800–2006. Weather, 62(4), 87–93. https://doi.org/10.1002/wea.67

McKee, T. B., Doesken, N. J., & Kleist, J. (1993). The relationship of drought frequency and duration to time scales. Proceedings of the 8th Conference on Applied Climatology, 17(22), 179–183.

Merz, B., Vorogushyn, S., Uhlemann, S., Delgado, J., & Hundecha, Y. (2012). HESS Opinions “More efforts and scientific rigour are needed to attribute trends in flood time series.” Hydrology and Earth System Sciences, 16(5), 1379–1387. https://doi.org/10.5194/hess-16-1379-2012

Mishra, A. K., & Singh, V. P. (2010). A review of drought concepts. Journal of Hydrology, 391(1), 202–216. https://doi.org/10.1016/j.jhydrol.2010.07.012

Murphy, C., Harrigan, S., Hall, J., Wilby, R. (2013). Climate-driven trends in mean and high flows from a network of reference stations in Ireland. Hydrological Sciences Journal, 58, 755 – 772. http://dx.doi.org/10.1080/02626667.2013.782407

O’Connell, P. E., Ewen, J., O’Donnell, G., & Quinn, P. (2007). Is there a link between agricultural land-use management and flooding? Hydrology and Earth System Sciences, 11(1), 96–107. https://doi.org/10.5194/hess-11-96-2007

O’Connor, P., Murphy, C., Matthews, T., & Wilby, R. L. (2021). Reconstructed monthly river flows for Irish catchments 1766–2016. Geoscience Data Journal, 8(1), 34–54. https://doi.org/10.1002/gdj3.107

Parry, S., Marsh, T., & Kendon, M. (2013). 2012: from drought to floods in England and Wales. Weather, 68(10), 268–274. https://doi.org/10.1002/wea.2152

Parry, S., Prudhomme, C., Wilby, R. L., & Wood, P. J. (2016a). Drought termination: Concept and characterisation. Progress in Physical Geography: Earth and Environment, 40(6), 743–767. https://doi.org/10.1177/0309133316652801

Parry, S., Wilby, R. L., Prudhomme, C., & Wood, P. J. (2016b). A systematic assessment of drought termination in the United Kingdom. Hydrology and Earth System Sciences, 20(10), 4265–4281. https://doi.org/10.5194/hess-20-4265-2016

Parry, S., Wilby, R., Prudhomme, C., Wood, P., & McKenzie, A. (2018). Demonstrating the utility of a drought termination framework: prospects for groundwater level recovery in England and Wales in 2018 or beyond. Environmental Research Letters, 13(6), 064040. https://doi.org/10.1088/1748-9326/aac78c

Parry et al. 2022. Dry summer pushes river flows to the brink of the 1976 drought. UKCEH Blog, August 2022. https://www.ceh.ac.uk/news-and-media/blogs/dry-summer-pushes-river-flows-brink-1976-drought

Parsons, D. J., Rey, D., Tanguy, M., & Holman, I. P. (2019). Regional variations in the link between drought indices and reported agricultural impacts of drought. Agricultural Systems, 173, 119–129. https://doi.org/10.1016/j.agsy.2019.02.015

Peña-Angulo, D., Vicente-Serrano, S. M., Domínguez-Castro, F., Lorenzo-Lacruz, J., Murphy, C., Hannaford, J., Allan, R. P., Tramblay, Y., Reig-Gracia, F., & el Kenawy, A. (2022). The Complex and Spatially Diverse Patterns of Hydrological Droughts Across Europe. Water Resources Research, 58(4), e2022WR031976. https://doi.org/10.1029/2022WR031976

Prosdocimi, I., Stewart, E. J., & Vesuviano, G. (2017). A depth–duration–frequency analysis for short-duration rainfall events in England and Wales. Hydrology Research, 48(6), 1624–1638. https://doi.org/10.2166/nh.2017.140

Rameshwaran, P., Bell, V., Brown, M.J., Davies, H., Kay, A.L., Rudd, A.C., Sefton, C. 2021. Use of Abstraction and Discharge Data to Improve the Performance of a National-Scale Hydrological Model. Water Resources Research, 58, e2021WR029787. https://doi.org/10.1029/2021WR029787

Rahiz, M., & New, M. (2012). Spatial coherence of meteorological droughts in the UK since 1914. Area, 44(4), 400–410. https://doi.org/10.1111/j.1475-4762.2012.01131.x

Robinson, M., Rodda, J. C., & Sutcliffe, J. (2013). Long-term environmental monitoring in the UK: origins and achievements of the Plynlimon catchment study. Transactions of the Institute of British Geographers, 38(3), 451–463. https://doi.org/10.1111/j.1475-5661.2012.00534.x

Rudd, A. C., Bell, V. A., & Kay, A. L. (2017). National-scale analysis of simulated hydrological droughts (1891–2015). Journal of Hydrology, 550, 368–385. https://doi.org/10.1016/j.jhydrol.2017.05.018

Rudd, A. C., Kay, A. L., & Bell, V. A. (2019). National-scale analysis of future river flow and soil moisture droughts: potential changes in drought characteristics. Climatic Change, 156(3), 323–340. https://doi.org/10.1007/s10584-019-02528-0

Rust, W., Bloomfield, J. P., Cuthbert, M., Corstanje, R., & Holman, I. (2022). The importance of non-stationary multiannual periodicities in the North Atlantic Oscillation index for forecasting water resource drought. Hydrology and Earth System Sciences, 26(9), 2449–2467. https://doi.org/10.5194/hess-26-2449-2022

Saft, M., Peel, M. C., Western, A. W., & Zhang, L. (2016). Predicting shifts in rainfall-runoff partitioning during multiyear drought: Roles of dry period and catchment characteristics. Water Resources Research, 52(12), 9290–9305. https://doi.org/10.1002/2016WR019525

Salwey, S., Coxon, G., Pianosi, F., Singer, M. B., & Hutton, C. (2023). National-scale detection of reservoir impacts through hydrological signatures. Water Resources Research, 59, e2022WR033893. https://doi.org/10.1029/2022WR033893

Schaller, N., Kay, A. L., Lamb, R., Massey, N. R., van Oldenborgh, G. J., Otto, F. E. L., Sparrow, S. N., Vautard, R., Yiou, P., Ashpole, I., Bowery, A., Crooks, S. M., Haustein, K., Huntingford, C., Ingram, W. J., Jones, R. G., Legg, T., Miller, J., Skeggs, J., … Allen, M. R. (2016). Human influence on climate in the 2014 southern England winter floods and their impacts. Nature Climate Change, 6(6), 627–634. https://doi.org/10.1038/nclimate2927

Sefton, C., Muchan, K., Parry, S., Matthews, B., Barker, L. J., Turner, S., & Hannaford, J. (2021). The 2019/2020 floods in the UK: a hydrological appraisal. Weather, 76(12), 378–384. https://doi.org/10.1002/wea.3993

Shorthouse, C., & Arnell, N. (1999). The effects of climatic variability on spatial characteristics of European river flows. Physics and Chemistry of the Earth, Part B: Hydrology, Oceans and Atmosphere, 24(1), 7–13. https://doi.org/10.1016/S1464-1909(98)00003-3

Shukla, S., & Wood, A. W. (2008). Use of a standardized runoff index for characterizing hydrologic drought. Geophysical Research Letters, 35(2). https://doi.org/10.1029/2007GL032487

Slater, L. J., Anderson, B., Buechel, M., Dadson, S., Han, S., Harrigan, S., Kelder, T., Kowal, K., Lees, T., Matthews, T., Murphy, C., & Wilby, R. L. (2020). Nonstationary weather and water extremes: a review of methods for their detection, attribution, and management. Hydrol. Earth Syst. Sci. Discuss., 2020, 1–54. https://doi.org/10.5194/hess-2020-576

Smith, K. A., Barker, L. J., Tanguy, M., Parry, S., Harrigan, S., Legg, T. P., Prudhomme, C., & Hannaford, J. (2019). A multi-objective ensemble approach to hydrological modelling in the UK: an application to historic drought reconstruction. Hydrology and Earth System Sciences. https://doi.org/10.5194/hess-23-3247-2019

Spraggs, G., Peaver, L., Jones, P., & Ede, P. (2015). Re-construction of historic drought in the Anglian Region (UK) over the period 1798–2010 and the implications for water resources and drought management. Journal of Hydrology, 526, 231–252. https://doi.org/10.1016/j.jhydrol.2015.01.015

Stahl, K., Hisdal, H., Hannaford, J., Tallaksen, L. M., van Lanen, H. A. J., Sauquet, E., Demuth, S., Fendekova, M., & Jódar, J. (2010). Streamflow trends in Europe: evidence from a dataset of near-natural catchments. Hydrology and Earth System Sciences, 14(12), 2367–2382. https://doi.org/10.5194/hess-14-2367-2010

Stahl, K., Tallaksen, L. M., Hannaford, J., & van Lanen, H. A. J. (2012). Filling the white space on maps of European runoff trends: estimates from a multi-model ensemble. Hydrol. Earth Syst. Sci., 16(7), 2035–2047. https://doi.org/10.5194/hess-16-2035-2012

Stahl, K., Kohn, I., Blauhut, V., Urquijo, J., de Stefano, L., Acácio, V., Dias, S., Stagge, J. H., Tallaksen, L. M., Kampragou, E., van Loon, A. F., Barker, L. J., Melsen, L. A., Bifulco, C., Musolino, D., de Carli, A., Massarutto, A., Assimacopoulos, D., & van Lanen, H. A. J. (2016). Impacts of European drought events: Insights from an international database of text-based reports. Natural Hazards and Earth System Sciences. https://doi.org/10.5194/nhess-16-801-2016

Stratford, C., Miller, J., House, A., Old, G., Acreman, M., Duenas-Lopez, M. A., Nisbet, T., Burgess-Gamble, L., Chappell, N., & Clarke, S. (2017). Do trees in UK-relevant river catchments influence fluvial flood peaks?: a systematic review. Centre for Ecology & Hydrology, Wallingford, UK. https://nora.nerc.ac.uk/id/eprint/517804/

Svensson, C., Hannaford, J., & Prosdocimi, I. (2016). Statistical distributions for monthly aggregations of precipitation and streamflow in drought indicator applications. Water Resources Research. https://doi.org/10.1002/2016WR019276

Svensson, C., & Hannaford, J. (2019). Oceanic conditions associated with Euro-Atlantic high pressure and UK drought. Environmental Research Communications, 1(10), 101001. https://doi.org/10.1088/2515-7620/ab42f7

Tallaksen, L. M., & van Lanen, H. A. J. (2004). Hydrological drought: processes and estimation methods for streamflow and groundwater (Vol. 48). Elsevier.

Tallaksen, L. M., Hisdal, H., & Lanen, H. A. J. van. (2009). Space–time modelling of catchment scale drought characteristics. Journal of Hydrology, 375(3), 363–372. https://doi.org/10.1016/j.jhydrol.2009.06.032

Tanguy, M., Chevturi, A., Marchant, B., MacKay, J.D., Parry, S. Hannaford, J., (2023) How will climate change affect spatial coherence of droughts in Great Britain? Environmental Research Letters, 18, 064048. DOI 10.1088/1748-9326/acd655

Tanguy, M., Prudhomme, C., Smith, K., & Hannaford, J. (2018). Historical gridded reconstruction of potential evapotranspiration for the UK. Earth System Science Data, 10(2), 951–968. https://doi.org/10.5194/essd-10-951-2018

Tanguy, M., Haslinger, K., Svensson, C., Parry, S., Barker, L. J., Hannaford, J., & Prudhomme, C. (2021). Regional Differences in Spatiotemporal Drought Characteristics in

Great Britain. Frontiers in Environmental Science, 9. https://doi.org/10.3389/fenvs.2021.639649

Tijdeman, E., Hannaford, J., & Stahl, K. (2018). Human influences on streamflow drought characteristics in England and Wales. Hydrology and Earth System Sciences, 22(2), 1051–1064. https://doi.org/10.5194/hess-22-1051-2018

Tijdeman, E., Stahl, K., & Tallaksen, L. M. (2020). Drought Characteristics Derived Based on the Standardized Streamflow Index: A Large Sample Comparison for Parametric and Nonparametric Methods. Water Resources Research, 56(10), e2019WR026315. https://doi.org/10.1029/2019WR026315

Turner, S., Barker, L. J., Hannaford, J., Muchan, K., Parry, S., & Sefton, C. (2021). The 2018/2019 drought in the UK: a hydrological appraisal. Weather, 76(8), 248–253. https://doi.org/10.1002/WEA.4003

van Loon, A. F., Gleeson, T., Clark, J., van Dijk, A. I. J. M., Stahl, K., Hannaford, J., di Baldassarre, G., Teuling, A. J., Tallaksen, L. M., Uijlenhoet, R., Hannah, D. M., Sheffield, J., Svoboda, M., Verbeiren, B., Wagener, T., Rangecroft, S., Wanders, N., & van Lanen, H. A. J. (2016). Drought in the Anthropocene. Nature Geoscience, 9(2), 89–91. https://doi.org/10.1038/ngeo2646

van Loon, A; Gleeson, T; Clark, J; van Dijk, A; Stahl, K; Hannaford, J; Di Baldassarre, G; Teuling, A; Tallaksen, L; Uijlenhoet, R; Hannah, D; Sheffield, J; Svoboda, M; Verbeiren, B; Wagener, T; Rangecroft, S; Wanders, N; van Lanen, H. 2016. Drought in a human-modified world: reframing drought definitions, understanding and analysis approaches. Hydrology and Earth System Sciences. 20 (9), 3631 – 3650. https://doi.org/10.5194/hess-20-3631-2016

van Loon, A. F. (2015). Hydrological drought explained. WIREs Water, 2(4), 359–392. https://doi.org/10.1002/wat2.1085

van Loon, A. F., Rangecroft, S., Coxon, G., Werner, M., Wanders, N., di Baldassarre, G., Tijdeman, E., Bosman, M., Gleeson, T., Nauditt, A., Aghakouchak, A., Breña-Naranjo, J. A., Cenobio-Cruz, O., Costa, A. C., Fendekova, M., Jewitt, G., Kingston, D. G., Loft, J., Mager, S. M., … van Lanen, H. A. J. (2022). Streamflow droughts aggravated by human activities despite management. Environmental Research Letters, 17(4), 044059. https://doi.org/10.1088/1748-9326/ac5def

Yevjevich, V. M. (1967). Objective approach to definitions and investigations of continental hydrologic droughts, An. Hydrology Papers, 23, Colorado State University, Fort Collins, Colorado, USA.

Wagener, T., Dadson, S. J., Hannah, D. M., Coxon, G., Beven, K., Bloomfield, J. P., Buytaert, W., Cloke, H., Bates, P., Holden, J., Parry, L., Lamb, R., Chappell, N. A., Fry, M., & Old, G. (2021). Knowledge gaps in our perceptual model of Great Britain's hydrology. Hydrological Processes, 35(7), e14288. https://doi.org/10.1002/hyp.14288

Watts, G., Christierson, B. von, Hannaford, J., & Lonsdale, K. (2012). Testing the resilience of water supply systems to long droughts. Journal of Hydrology, 414–415, 255–267. https://doi.org/10.1016/j.jhydrol.2011.10.038

Watts, G., Battarbee, R. W., Bloomfield, J. P., Crossman, J., Daccache, A., Durance, I., Elliott, J. A., Garner, G., Hannaford, J., Hannah, D. M., Hess, T., Jackson, C. R., Kay, A. L., Kernan, M., Knox, J., Mackay, J., Monteith, D. T., Ormerod, S. J., Rance, J., … Wilby, R. L. (2015). Climate change and water in the UK – past changes and future prospects. Progress in Physical Geography: Earth and Environment, 39(1), 6–28. https://doi.org/10.1177/0309133314542957

West, H., Quinn, N., & Horswell, M. (2022). The influence of the North Atlantic oscillation & East Atlantic pattern on drought in British catchments. Frontiers in Environmental Science, 10, Article 754597. https://doi.org/10.3389/fenvs.2022.754597.

West, H., Quinn, N., & Horswell, M. (2022). Spatio-temporal propagation of North Atlantic Oscillation (NAO) rainfall deviations to streamflow in British catchments. Hydrological Sciences Journal, 67(5), 676–688. https://doi.org/10.1080/02626667.2022.2038791

Wilby, R.L. (2006). When and where might climate change be detectable in UK river flows? Geophysical Research Letters, 33, L19407, doi:10.1029/2006GL027552

Wilby, R. L., Clifford, N. J., de Luca, P., Harrigan, S., Hillier, J. K., Hodgkins, R., Johnson, M. F., Matthews, T. K. R., Murphy, C., Noone, S. J., Parry, S., Prudhomme, C., Rice, S. P., Slater, L. J., Smith, K. A., & Wood, P. J. (2017). The 'dirty dozen' of freshwater science: detecting then reconciling hydrological data biases and errors. WIREs Water, 4(3), e1209. https://doi.org/10.1002/wat2.1209

Wilhite, D. A., & Glantz, M. H. (1985). Understanding: the drought phenomenon: the role of definitions. Water International, 10(3), 111–120.

World Meteorological Organization, & Global Water Partnership. (2016). Integrated Drought Management Programme Handbook of Drought Indicators and Indices (M. Svoboda & B. A. Fuchs, Eds.; Integrated). Integrated Drought Management Programme. www.droughtmanagement.info

Vicente-Serrano, S. M., Domínguez-Castro, F., Murphy, C., Hannaford, J., Reig, F., Peña-Angulo, D., Tramblay, Y., Trigo, R. M., mac Donald, N., Luna, M. Y., Mc Carthy, M., van der Schrier, G., Turco, M., Camuffo, D., Noguera, I., García-Herrera, R., Becherini, F., della Valle, A., Tomas-Burguera, M., & el Kenawy, A. (2021). Long-term variability and trends in meteorological droughts in Western Europe (1851–2018). International Journal of Climatology, 41(S1), E690–E717. https://doi.org/10.1002/joc.6719

Zaidman, M. D., Rees, H. G., & Young, A. R. (2002). Spatio-temporal development of streamflow droughts in north-west Europe. Hydrology and Earth System Sciences, 6(4), 733–751. https://doi.org/10.5194/hess-6-733-2002

C: Modelling climate change impacts on UK hydrological drought: A review

Rosanna Lane and Alison Kay,

UK Centre for Ecology & Hydrology

Overview

This report summarises the state-of-the-art in modelling climate change impacts on river low flows and streamflow drought for the UK, focussing on 1) modelling approaches, 2) projections, 3) remaining knowledge gaps.

Approaches:

Most hydrological climate change impact studies follow a 'top-down' approach using a climate-hydrological modelling cascade. These are underpinned by climate model data, which have seen many improvements in recent years. Remaining issues include substantial biases in climate model precipitation, disparities between climate and hydrological model resolutions, and need for high-resolution, spatially and temporally consistent projections from a range of climate models. Hydrological modelling of national-scale climate change impacts also faces challenges, such as developing model structures/ parameters that are robust to non-stationarities, developing seamless parameter fields, and incorporating human-water interactions. A major challenge for both climate and hydrological modelling is dealing with the cascading uncertainties, ensuring that modelling uncertainties are adequately represented and communicated whilst also remaining practical for analysis.

Several alternative 'bottom-up' approaches to exploring climate change impacts have also emerged. These tend to focus on system sensitivity to change or the robustness of certain plans, and can be carried out independently from climate change scenarios. Examples include scenario-neutral response surfaces and alternative storylines of past events.

Projections:

Climate projections show nationwide increases in temperature, increases in summer potential evapotranspiration, and a reduction in summer rainfall; all factors that could result in declining flows. However, projections also indicate increased winter precipitation, heavier summer downpours and an increased chance of high intensity rainfall events, leading to a complex picture for changing river flows and hydrological droughts. Projections from hydrological modelling studies tend to show reductions in median flows, summer flows and low flows across the UK, although with large uncertainties and local variations. Most projections indicate an increase in the severity of future drought events, although changes to other drought characteristics such as intensity and duration vary regionally. Southeast England emerges as a hot-spot for future multi-year droughts, with increases in both drought intensity and duration.

Gaps:

A key issue is adequately representing uncertainties in national climate change impacts. It is important that uncertainties are represented in modelling studies to support informed decision-making. The evidence for national-scale changes to streamflow drought is largely based on the Hadley Centre Climate model, through the weather@home and UKCP18 Regional projections. This could be leading to a skewed view of UK drought impacts, as projections driven by other climate models show smaller changes to drought conditions, and even possible reductions in drought occurrence for northern England and Scotland. There is therefore a need for high-resolution climate projections from a broader range of climate models to underpin future hydrological modelling efforts. Additionally, most national-scale studies include a single hydrological model structure with a single set of model parameters. But hydrological modelling uncertainties can be very large for low flows, particularly in southeast England, and it is therefore important that these are also represented.

Other research gaps include: 1) there is a need for information on possible low-likelihood but high-impact events, to ensure adaptation measures are robust to worst-case scenarios; 2) there are relatively few studies evaluating climate change impact on UK droughts; studies with different driving data and hydrological models are needed to comprehensively understand changes to drought duration, intensity, frequency and spatial extent at the national scale; 3) further hydrological model development and evaluation is needed for models that can robustly simulate low flows under non-stationary conditions; 4) there is a need for evaluations of future changes in other environmental/anthropogenic factors, such as land-use or abstractions/discharges, in combination with climate change.

1. Introduction

Droughts are a recurrent natural hazard in the UK, with serious impacts that can cover extensive areas and persist for long durations (Van Loon, 2015). A drought can be defined as a sustained period of below normal moisture conditions and subsequent limitations in water availability (IPCC, 2021). Within this broad definition there are many different types of drought, including meteorological drought (unusually low rainfall), soil moisture drought (unusually low soil moisture), groundwater drought (unusually low groundwater levels) and streamflow drought (unusually low river discharge). These different types are all interconnected, but this review will focus on streamflow drought and river low flows (different types of drought are covered in other reviews, e.g. Bloomfield and Jackson, 2023; Counsel and Durant, 2023). Streamflow droughts can have serious impacts spanning multiple sectors, including water supply, energy and industry (hydropower and cooling water), crop production, navigation and recreation (Van Loon, 2015).

Climate change could exacerbate drought risk for the UK, potentially leading to more intense and longer lasting droughts in the future (IPCC, 2021; Rudd et al., 2019; Smith et al., 2019). The latest climate projections for the UK indicate warmer, wetter winters, hotter, drier summers and an increase in the frequency and intensity of extreme events (Met Office, 2019). The impact of these meteorological changes on streamflow drought is complex, as streamflow response varies depending on antecedent conditions and catchment characteristics. Projections generally indicate a reduction in summer flows and more severe droughts, but there are large uncertainties in the magnitude of future changes (Kay et al., 2018; Lane and Kay, 2021; Rudd et al., 2019). To manage drought risk and build resilient water supply systems, it is crucial that we understand the characteristics of extreme drought events and how these may change in the future.

This review summarises the state-of-the-art in modelling climate change impacts on low flows and drought for the UK, highlighting limitations of current methods, gaps in our understanding, and opportunities for future research. It aims to address the following questions:

1. What approaches have been taken to model climate change impacts for low flows and drought, and what are the key data and methodological limitations with these approaches?

2. What are the projected impacts of climate change for low flows and droughts?

3. What do we still not know about climate change impacts for low flows and droughts, and how can we improve our understanding?

To answer these questions, the review has been split into 4 sections. Section 2 focusses on approaches to model climate change impacts, section 3 reviews the evidence for climate change impacts on low flows and droughts in the UK, and section 4 highlights remaining knowledge gaps.

2. Approaches to model climate change impacts

Most hydrological climate change impact assessments for the UK follow a scenario-led or 'top down' approach (e.g. Charlton and Arnell, 2014; Christierson et al., 2012; Dobson et al., 2020; Kay et al., 2021b, 2021a, 2018; Lane and Kay, 2021; Prudhomme et al., 2012; Rudd et al., 2019). This usually involves a climate-hydrological modelling chain where emissions scenarios are used as input to climate models and the subsequent climate projections are then used as input to hydrological models (Chan et al., 2022). There are many decisions in the modelling chain, including the choice of emissions scenarios, Global Climate Models (GCMs), downscaling method, hydrological models, model parameters, and river flow/drought metrics. Each of these choices will impact the subsequent flow projections (Wilby and Harris, 2006), leading to a 'cascade of uncertainty' with large uncertainty ranges in the resultant flow and drought projections (Figure 1).

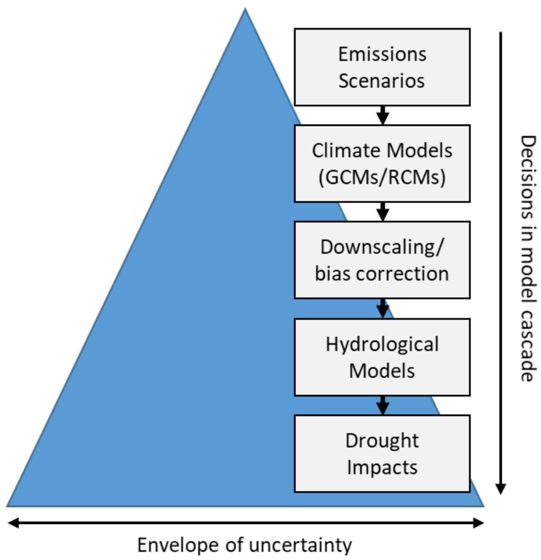


Figure 1. Overview of the cascade of uncertainty in GCM-driven climate impact assessments. Each decision made in the model cascade adds an element of uncertainty, resulting in a very wide envelope of uncertainty in the resultant flow/ drought impacts.

This section discusses the approaches used to model climate change impacts, focusing on current practice and remaining limitations. This includes the use of climate model data (Section 2.1), hydrological models (2.2), drought indicators (2.3), uncertainties in the model cascade (2.4) and alternative bottom-up approaches (2.5).

2.1. Using climate model data for hydrological applications

Approaches to using climate model data

Most climate impact analyses use data from global climate models (GCMs), often with nested higher resolution regional climate models (RCMs), to provide future climate projections. Recent years have seen many advancements to climate modelling capabilities including: higher resolution simulations which can better simulate spatial patterns and the seasonal cycle of rainfall; larger ensembles to capture climate variability and projection uncertainties; and better representation of land surface processes (Flato et al., 2013; Gutiérrez et al., 2021; Murphy et al., 2018). For example, the most recent UK Climate Projections (UKCP18) produced in 2018 advanced upon the 2009 product through (1) increased resolution of the GCM from ~300 to ~60km, (2) increased resolution of the RCM from 25 to 12km, (3) inclusion of very high resolution (2.2km) projections, (4) improved model processes and parameterisations (Murphy et al., 2018).

However, despite these improvements there are still substantial biases in climate model outputs, and models continue to perform less well for precipitation than for temperature (Flato et al., 2013; Gutiérrez et al., 2021). Some studies have applied climate model output directly, assuming that the changes between baseline and future climatology are meaningful despite biases (Bell et al., 2007). However, most climate impact analyses apply some form of bias correction (BC), especially for precipitation. There are a range of BC methods available, from simple monthly-mean corrections to more complex quantile-mapping approaches, with no consensus for best practice (Addor and Seibert, 2014; Maraun et al., 2017; Teutschbein and Seibert, 2012).

BC approaches generally improve the ability of the climate-hydrological model cascade to match flow statistics during the baseline period, but many issues have been identified (Maraun et al., 2017). Limitations include; 1) BC cannot improve credibility of climate model outputs, and could even hide a lack of credibility, 2) BC assumes that model biases are nonstationary and therefore the same correction is suitable for baseline and future periods, 3) BC relies on observational reference datasets, which have uncertainties especially if they have been interpolated into a gridded product, 4) BC can impair the relationship between RCM output variables, and alter the spatiotemporal field consistency, 5) BC usually only corrects for biases in one statistic – e.g. correction of daily precipitation distributions may not improve distributions of multi-day precipitation (Addor and Seibert, 2014; Ehret et al., 2012; Maraun et al., 2017; Muerth et al., 2013). The choice of BC method is a relatively small source of uncertainty compared to climate model selection, but can still impact the magnitude of future flow changes (Muerth et al., 2013; Smith et al., 2014; Tian et al., 2013). Maraun et al., (2017) recommends that BC should be carried out based on solid knowledge of the relevant climatic phenomena and the ability of GCMs to simulate them, calling for closer collaborations between disciplines.

There is still a disparity in resolution between climate model output and typical hydrological model inputs, although new convection-permitting climate model runs are helping to resolve this (Kay, 2022; Kendon et al., 2019). GCM resolutions (~60km+) remain too coarse for direct use in hydrological modelling, which usually requires data closer to a 1km resolution. Therefore downscaling approaches are often applied, including dynamical downscaling (e.g. using a nested RCM) or statistical downscaling (post-processing GCM/RCM output). A simple method of downscaling precipitation based on patterns of standard-period average annual rainfall (SAAR) has been applied in many UK-based climate impact studies (see Table 1), and has recently been shown to perform well (Kay et al., 2023)(Kay et al., 2023). With RCM output being available at increasingly high resolutions, some studies do not apply any further downscaling. For example, Lane et al. (2022) used 12km RCM output, but limited their analysis to flows in larger (>144km2) catchments. Again, the choice of downscaling approach adds a further source of uncertainty into the climate-hydrological modelling chain, with no consensus on a best method.

An alternative approach to dealing with issues of biases and coarse resolutions in climate model data is the delta-change method. This approach extracts changes in meteorological variables from the climate model data and uses these to perturb observed time-series to drive a hydrological model, rather than the direct forcing approach which uses time-series of climate model data to drive a hydrological model (after any bias correction or downscaling has been applied). The perturbation is often based on monthly means (e.g. Kay et al., 2021b), but could also be based on other statistics such as quantiles (e.g. Smith et al., 2014). The delta change technique produces data at the spatial and temporal resolution required for hydrological modelling, as it is based on observational data. It also implicitly accounts for any biases in the climate model data by focusing only on changes in meteorological variables (although some studies correct for biases before calculating these changes). However, only the direct forcing method can capture changes in the spatial pattern or temporal sequencing of events, as the delta-change approach is limited by the events in the observed time-series. The delta-change approach is also heavily reliant on the chosen statistic, for example a study using monthly-mean precipitation changes could miss very large increases in extreme precipitation.

Hydrological model simulations are usually carried out independently ('offline') from climate model simulations. This means that there may be inconsistencies between the climate and hydrological simulations, for example soil moisture levels may be inconsistent, and it is not possible to include any feedbacks between the hydrological and climate models. The recent increase in resolution of coupled climate models means that they could be used to directly simulate river flows.

Available climate data for the UK

A range of climate projections are available, and have been used to evaluate climate change impacts on low flows/drought, for the UK (See Table 1). Many impact studies were based on the UK Climate Projections produced in 2009 (UKCP09), which included a perturbed-parameter ensemble (PPE) of RCM projections (~25km), probabilistic projections, and weather-generator time-series (Charlton and Arnell, 2014; Christierson et al., 2012; Kay and Jones, 2012; Rahiz and New, 2013). A series of studies used a large ensemble of 100 time-series for three time periods generated from the weather@home system, part of the wider climateprediction.net project, to evaluate changing low flows and droughts (Dobson et al., 2020; Guillod et al., 2018; Kay et al., 2018; Rudd et al., 2019). Most recently, the UKCP18 projections, which include global projections from a range of GCMs, a perturbed parameter RCM ensemble (~12km) and probabilistic projections, have been used in a handful of studies (Arnell et al., 2021b; Kay et al., 2021b, 2021a; Lane and Kay, 2021). A national dataset of hydrological projections based on the UKCP18 RCM ensemble have also been made available (Hannaford et al., 2022)(Hannaford et al., 2022; Parry et al., 2023). The different types of available climate data each have their own advantages/limitations, and a combination is recommended to fully understand the range of possible impacts (Kay and Jones, 2012).

A useful feature of the UK Climate Projections is that they include the characterisation of climate model uncertainties. Both UKCP09 and UKCP18 include probabilistic projections based on a number of different GCMs, as well as a perturbed parameter RCM ensemble based on the Hadley Centre Model (Murphy et al., 2018). A few studies have used the probabilistic climate projections to assess climate change impacts for river flows (Christierson et al., 2012; Kay et al., 2020). However, while the probabilistic projections are useful to capture climate uncertainties, they are not spatially coherent, so cannot be used to capture widespread events or be used directly as hydrological model input. The UKCP18 RCM output is relatively high resolution (12km), and spatially and temporally coherent, and therefore is more commonly used for hydrological modelling applications (e.g. Hannaford et al., 2022; Kay et al., 2021a; Lane et al., 2022; Lane and Kay, 2021). A key advantage of RCM projections is that they can be used to simulate widespread events, looking at the changing spatial extent and coincidence of droughts. They also indicate climate modelling uncertainties through the use of a PPE, albeit an underestimation of total climate uncertainty as all ensemble members are based on a single GCM structure and emissions scenario.

Table 1. Summary of modelling decisions in selected studies evaluating climate change impacts on low flows and droughts across the UK.

Reference	Spatial coverage	Climate product	Climate data processing	Hydrological Model(s)	Impacts evaluated
New et al., (2007)	Thames, UK	climateprediction.net probabilistic (x449)	Seasonal-mean climate data linearly interpolated to monthly change factors, delta-change approach	CATCHMOD	Q05, Q50, Q95
Lopez et al., (2009)	Wimblebal l water resource zone, SW England	climateprediction.net PPE (x246), CMIP3 (x21 GCMs)	Monthly climate data, quantile-quantile transform method (delta-	CATCHMOD and LANCMOD water resources	N month deficits, ability to satisfy

			change approach)	system model	water demand
Cloke et al., (2010)	River Medway, SE England	UKCP09 RCM PPE subset (x10)	Distribution based scaling bias correction tested, but uncorrected RCM output used	CATCHMOD	Flow quantiles (including Q95, Q50, Q05), monthly/ seasonal flows
Christierson et al. (2012)	UK – 70 catchments	UKCP09 probabilistic (x20, sampled from 10,000)	Delta-change approach	PDM and CATCHMOD	Mean annual flow, monthly flow
Prudhomme et al. (2012)	Great Britain – catchment modelling interpolated to 1km grid	UKCP09 RCM PPE (x11)	Delta-change approach	CERF	Seasonal mean flows
Charlton and Arnell (2014)	England - 6 catchments	UKCP09 probabilistic (x10,000)	Delta-change approach	Cat-PDM	Average annual runoff, Q5, Q50 and Q95
Kay et al. (2018)	Great Britain – 1km grid	weather@home (x100)	Monthly change factor bias-correction and SAAR downscaling for precipitation	G2G	2- and 20-year return period flows.
Rudd et al. (2019)	Great Britain – 1km grid	weather@home (x100)	Monthly change factor bias-correction and SAAR	G2G	Standardised drought intensity, duration

			downscaling for precipitation		and severity.
Dobson et al. (2020)	England and Wales – 80 catchments	weather@ home (x100)	Monthly change factor bias-correction and SAAR downscaling for precipitation	DECIPHeR (and water resource system modelling)	Standardized indices for precipitation, streamflow and reservoir storage volume. Drought Coincidence Index.
Kay et al., (2021b)	Great Britain – 1km grid	UKCP18 Global (x28), UKCP18 Regional (x12)	Delta-change approach	G2G	5- and 20-year return period low flows
Lane and Kay (2021)	Great Britain – 1km grid	UKCP18 Regional (x12)	Monthly change factor bias-correction and SAAR downscaling for precipitation.	G2G	10-year return period low flows
Hannaford et al. (2022); **Parry et al., (2023); Tanguy et al.,submitted)**	UK – 200 catchments	UKCP18 Regional (x12)	Monthly change factor bias-correction and SAAR downscaling for precipitation.	G2G, PDM, GR4J and GR6J	in progress

2.2. Hydrological modelling for climate change impacts

Hydrological models are an important tool for transforming meteorological variables into river flow, taking into account the relevant catchment/landscape characteristics. There are a wide variety of hydrological models available, and that have been used for climate change impact analyses in the UK (e.g. CATCHMOD - Cloke et al., 2010; Lopez et al., 2009; New et al., 2007; Grid-to-Grid - Bell et al., 2012, 2009; Rudd et al., 2019; DECIPHeR - Coxon et al., 2019; Lane et al., 2022; HBV - Bergström, 1992; Smith et al., 2014; GR4J - Perrin et al., 2003; Tian et al., 2013). These vary in complexity, resolution and process representation, ranging from simple lumped catchment-based models to high-resolution gridded physically-based models. Importantly there is no overall best hydrological model, as different models will be fit-for-purpose for different applications (Horton et al., 2022).

Challenges for hydrological modelling of climate change impacts on droughts include:

Non-stationarities

Hydrological models are often developed and calibrated with the assumption that climatic conditions will not change over time, yet climate change analyses require models to use non-stationary climate data. Alongside changing climate, extreme hydrological events can profoundly change the nature of the hydrological system itself (Fowler et al., 2022). For example, in the context of the Australian Millennium drought, Fowler et al., (2022) found that some watersheds experienced a ‘shift’ in hydrological behaviour that models could not replicate – leading to models under-predicting drought impacts. In the context of climate change, shifting behaviour could amplify the severity of future droughts beyond what is predicted by our models. Allowing for non-stationarities in model parameters and physics therefore remains an important issue (Slater et al., 2021). Usually, model performance is evaluated over a baseline period, as that is when flow data is available (e.g. Kay et al., 2021a; Lane et al., 2022; Steele-Dunne et al., 2008). However, good model performance under the current climate (which models are calibrated and developed for) does not necessarily mean good performance under different climate conditions. It is therefore important to develop model structures and calibration procedures that can increase robustness to non-stationarities (Brigode et al., 2013).

Differential split sample tests (DSSTs) can be used to evaluate the robustness of hydrological model calibration to non-stationary climatic inputs (Coron et al., 2012; Klemes, 1986; Seiller et al., 2012). DSSTs select periods in the observational record with distinct climate characteristics (e.g. wetter/drier periods). The hydrological model is then calibrated and validated for periods with different climate characteristics, to show how the model performs under a climate that differs from the calibration period. This can give useful insights into potential model biases when simulating climate change. For example, Coron et al., (2012) found that calibration over a wetter (drier) catchment led to an overestimation (underestimation) of mean simulated runoff, concluding that changing climate could introduce significant errors in simulations. Seiller et al., (2012) carried out a DSST using 20 lumped conceptual models, finding that while some model structures offered a good compromise in terms of model performance and robustness to change, others could provide misleading results under a non-stationary climate. Trotter et al.,

(2023) focused on how well models calibrated for normal conditions could simulate the Australian Millennium drought, finding extensive degradation of model performance, mainly due to over-estimation of flow, during the drought. Evaluation of hydrological models using a DSST is not common practice, but could aid selection of model structures/ parameter sets that are more robust to non-stationarities.

Model structure and parameter selection:

The choice of hydrological model structure and parameters can have a large effect on low flow simulation and subsequent climate change impacts, and it is therefore good practice to include multiple hydrological models and/or parameter sets within climate impact assessments (Lane et al., 2022; Mendoza et al., 2015; Seiller et al., 2012). Often different hydrological model structures or parameter sets have similar performance in a baseline simulation, but then simulate different flow responses to climatic changes (Karlsson et al., 2016; Mendoza et al., 2016, 2015). Inclusion of a single hydrological model structure and parameterisation may therefore give an overconfident and potentially misleading portrayal of climate change impacts. Including multiple hydrological models/parameter sets allows for a more robust assessment (Seiller et al., 2012), and can highlight areas where hydrological modelling uncertainties are large (e.g. Lane et al., 2022, Parry et al. in prep).

However, despite the benefits, there are few national climate impact assessments that use multiple hydrological model structures / parameterisations for the UK. Difficulties include; the already large computational demand when running national-scale models, varying data requirements, and high barriers to implementation of a new hydrological model. Ways to overcome these difficulties include; a) collaborative projects, b) multi-model frameworks, c) parameter uncertainty frameworks. For example, the recent eFLaG project involved collaboration between multiple research centres and groups within UKCEH, producing a national dataset of future river flow and groundwater projections from four hydrological and two groundwater models (Hannaford et al., 2022). Multi-model frameworks allow users to easily switch between hydrological model structures, within a common modelling infrastructure. Examples include the Framework for Understanding Structural Errors (FUSE - Clark et al., 2008), the Modular Assessment of Rainfall-Runoff Models Toolbox (MARRMoT - Knoben et al., 2019) and UniFHy (Hallouin et al., 2021).

Regional parameterisation/ calibration:

To capture large-scale droughts and changes in drought extent, we need hydrological models that can run in a spatially coherent way across regional to national domains. A key challenge for this is model parameterisation, and specifically the generation of spatial parameter fields. Usually, catchment-based hydrological models are calibrated against river flows at the catchment outlet. When scaling up to the national level, individually calibrating models to each catchment can lead to spatial discontinuities between catchments, with a ‘patchwork quilt’ effect of parameter fields with unrealistic differences at catchment boundaries (Mizukami et al., 2017). A potential way forward is the multiscale parameter regionalisation (MPR) technique, which relates parameters to geophysical data to create seamless parameter fields (Mizukami et al., 2017; Samaniego et al., 2017, 2010). The MPR technique has been used within the mesoscale Hydrologic Model (mHM) to evaluate changes in low flows across Europe (Marx et al., 2018). It has also recently been implemented for the DECIPHeR model across Great Britain (Lane et al., 2021), and used to evaluate climate change impacts on high flows (Lane et al., 2022), but has not yet been used for low flow or drought impacts.

Spatial variability in model performance

The ability of hydrological models to reproduce river flows varies across the country. Lane et al., (2019) evaluated the performance of four lumped hydrological models across Great Britain, finding a general pattern of good model performance for the wetter catchments to the west, and poor model performance in south-east England. Other model evaluations have found similar patterns, with large biases for groundwater dominated catchments in the south-east (Coxon et al., 2019). This has implications for our understanding of future droughts; the south-east is a hotspot for future droughts but is also often where are hydrological model predictions are poorest and have the largest uncertainties (Lane et al., 2022, 2019).

Representation of human-water interactions:

Human-water interactions (e.g. abstractions, discharges, water transfers, reservoir operation) can ameliorate/ worsen drought impacts, but are not currently included in most hydrological models for the UK. For example, the commonly applied national-scale Grid-to-Grid model usually simulates natural river flows (Bell et al., 2009), the DECIPHeR model does not yet include human influences at the national scale (Coxon et al., 2019), and for lumped catchment models these processes are generally accounted for by calibration rather than being explicitly included in the model structure. Incorporation of abstraction/discharge data into the Grid-to-Grid model has been shown to generally improve simulation of river flows, especially low flows, but data availability remains a key barrier (Rameshwaran et al., 2022). Furthermore, these national / gridded models usually have a relatively limited representation of groundwater, so representation of abstraction in groundwater dominated catchments can have large simulation errors.

2.3. From streamflow to droughts

It is important that a distinction is made between low flows and drought events. Low flows refer to low river discharge. Evaluations of changing low flows often use annual minimum time series, which will not reflect a streamflow drought in all years (Van Loon, 2015). In contrast, droughts refer to below-normal river flow conditions, often across large areas and for long durations, and can be described in terms of event timing, duration, severity, intensity and/or spatial extent.

Droughts are difficult to define quantitatively, due to their slow onset, slow recovery and range of impacted sectors, and this has led to a variety of drought indices being developed (Van Loon, 2015). There is no consensus on a best index, but the selected index is important as different indices could result in different conclusions, especially in a climate change impact context.

The main approaches include; 1) standardised indices such as the standardised precipitation index (SPI) or standardised streamflow index (SSI), or 2) threshold methods using time-series of streamflow. The standardised indices represent anomalies from normal conditions in a standardised way, enabling regional analysis. For example, the SSI fits a distribution to an accumulation of streamflow over a given period to provide a normalised value (e.g. Dobson et al., 2020). Threshold methods define drought periods as when time-series of hydrometeorological variables are below a pre-defined threshold level (Van Loon, 2015). A key advantage of the threshold method is that it can give deficit volume (i.e. the 'lacking' volume of water below the threshold) which is important for water resources management. However, results will be sensitive to the chosen threshold value.

Droughts can be described in terms of duration (length of time flows are below threshold), intensity (magnitude below threshold), and severity (duration multiplied by mean intensity, also referred to as accumulated deficit). These are illustrated in Figure 2, using a threshold of SSI<1.5 to define drought events. Drought metrics are covered in other reviews (Hannaford et al., 2023), so will not be discussed further here.

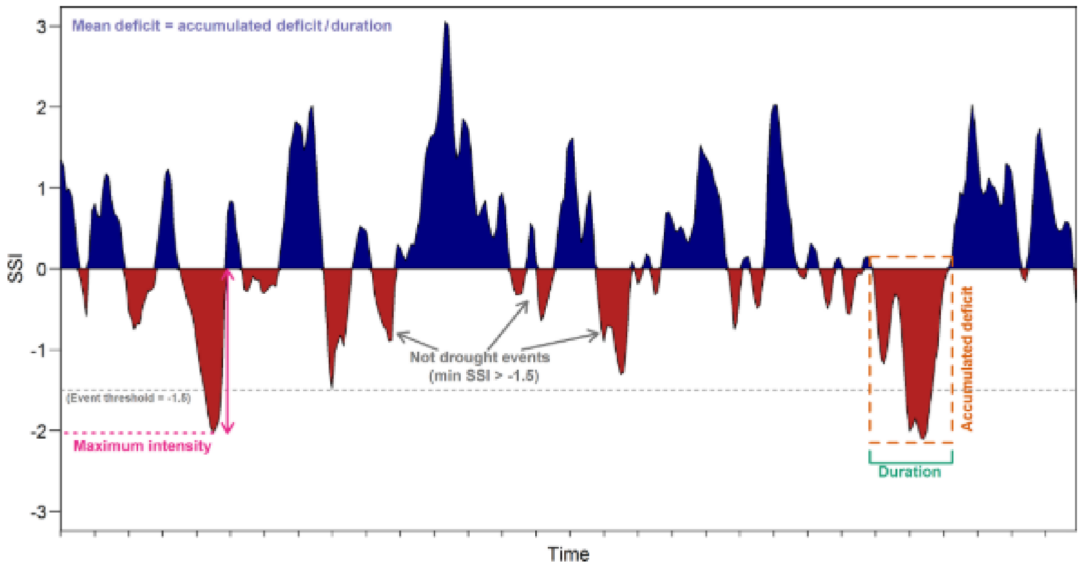


Figure 2. Conceptual diagram illustrating drought event identification and characteristics, reproduced from Barker et al., (2019).

2.4. Uncertainties

Large uncertainties are a widely recognised issue for climate impact assessments. Many studies have quantified the importance of different uncertainty sources, through carrying out climate impact assessments with a range of methodological choices (e.g. different GCMs / RCMs / downscaling approaches / hydrological models). These generally find that future climate is the largest source of uncertainty, with GCM selection usually the dominant uncertainty source and emissions scenario becoming increasingly important over time (Addor et al., 2014; Bastola et al., 2011; Chegwidden et al., 2019; Kay et al., 2009; Vetter et al., 2017). It is now common practice to include a GCM/RCM ensemble to represent these uncertainties.

However, hydrological models can be an important source of uncertainty for low flow impacts. Studies across Europe and North America have found that the spread in projections resulting from hydrological model choice can be larger than that from GCM choice for some regions (Chegwidden et al., 2019; De Niel et al., 2019; Hagemann et al., 2013; Meresa and Romanowicz, 2017; Velázquez et al., 2013; Vetter et al., 2017). Mendoza et al., (2016) found that hydrological model structure and parameter estimation could impact the direction and magnitude of projected changes in the annual water balance for catchments in the Colorado River Basin. However, inclusion of hydrological modelling uncertainties in national-scale simulations for the UK is still rare. Notable exceptions are an application of the DECIPHeR model with 30 different parameter sets to investigate changing high flows across Great Britain (Lane et al., 2022), and the eFLaG projections of climate change impacts on river flows from four hydrological models (Hannaford et al., 2022; Parry et al., in prep.).

The relative contribution of different uncertainty sources varies between catchments with different characteristics (Addor et al., 2014; Mendoza et al., 2015; Yang et al., 2019). For example, Lane et al. (2022) find hydrological model parameter uncertainties to be particularly large for many catchments in southeast England. These catchments tend to be a) relatively dry compared to the rest of the country, with lower precipitation and higher PET, b) overlaying chalk aquifers leading to possible groundwater transfers between catchments and c) in an area of high population density with abstractions/returns and other management impacts river flows. Interactions between different components of the modelling chain can also be an appreciable source of uncertainty (Aryal et al., 2018; De Niel et al., 2019).

It is important that uncertainties are adequately represented, to prevent biased or misleading conclusions and to support informed decision making (Smith et al., 2018). Inclusion of multiple uncertainty sources enables identification of robust regime changes that emerge despite projection uncertainty (Addor et al. 2014). However, uncertainty can never be quantified comprehensively, so at best we have a partial representation of the full range of uncertainties in future drought projections.

2.5. Alternative bottom-up approaches

Bottom-up approaches to explore climate change impacts have been developed as an alternative to 'top-down' approaches that use a climate-hydrological modelling chain. Bottom-up approaches can be carried out independently from climate change scenarios (often termed scenario-neutral), with a greater focus on catchment sensitivity to climatic changes or the robustness of certain plans over time (Chan et al., 2022). These can consider a wider range of possible futures, including low likelihood but high impact events which may not be fully represented in GCM projections, thus complementing GCM-driven approaches.

One example is the scenario-neutral response surface technique, which has been applied to evaluate climate change impact on flood peaks across the UK (Kay et al., 2021c, 2014; Prudhomme et al., 2013, 2010) and to low flows for two British catchments (Prudhomme et al., 2015). Response surfaces characterising catchment sensitivity to climatic change are generated through hundreds of model runs with incremental changes in temperature/precipitation characteristics (Kay et al., 2014; Prudhomme et al., 2015, 2010). This enables visualisation of catchment response to a wide range of possible outcomes, and can also be used in combination with climate projections by overlaying projections on the response surface. Prudhomme et al. (2015) demonstrate that this approach can be of great value in understanding future pressures on water resources. Key advantages include 1) the shape of the response surface highlights changes to which the system is most sensitive, enabling management responses that are robust to a wide range of plausible climatic changes, 2) understanding system sensitivity can also highlight extremes that are possible within current climate variability, 3) new projections can be mapped onto the response surfaces when they become available, preventing the need to replicate hydrological modelling efforts.

However, an important limitation of the response surface technique is the unavoidable simplification of complex processes. It is generally limited to visualising two dimensions at a time, whereas flow changes will be dependent on a complex range of factors, with additional response surfaces required to visualise different variables. Quinn et al., (2020) also show that decisions made during the design of the scenario-neutral response surface technique can influence which policies appear most robust for preventing future water scarcity. For example, deciding on the range of changes in temperature and precipitation to show on response surfaces, and whether these ranges should be conditioned on historical data, paleo data, or climate projections, could impact the robustness rank of different policies. They therefore recommend that vulnerability assessments should include competing hypotheses of how the future might evolve, with multiple possible experimental designs being considered to find policies that perform consistently well.

Another bottom-up approach is the use of storylines to visualise how observed events could have unfolded given different conditions. For example, Chan et al., (2022) explored storylines of the 2010-2012 UK drought, with hypothetical changes to precondition severity, temporal drought sequence and climate change. The storylines showed that most catchments would be vulnerable to a drought with a third dry year, and that climatic changes would point towards a worsening of drought conditions across most of the UK given a repeat of the 2010-2012 drought sequence. This provided a useful way to stress-test UK catchments against plausible, but yet unseen, extreme drought conditions.

3. Projections of climate change impacts

3.1. Climatic changes

The headline finding from the UKCP18 projections is for “an increased chance of warmer, wetter winters and hotter, drier summers along with an increase in the frequency and intensity of extremes” (Met Office, 2019). These findings are consistent across the UKCP18 range of projections over land, which include: probabilistic projections for the UK covering five emissions scenarios; global projections (60km) from the Hadley Centre GC3.05 and CMIP5 GCMs; a perturbed-parameter ensemble of the Hadley Centre RCM across Europe (12km); and a convection permitting model (CPM) run at high resolution (2.2km) across the UK nested in the RCM ensemble.

There is high confidence in projections of rising temperatures, with the latest IPCC AR6 report stating it was ‘virtually certain’ warming will continue across Europe irrespective of emissions scenario. For the UK, temperature projections indicate annual mean increases of around +1oC to +6oC by the end of the century (Murphy et al., 2018). Temperature increases vary spatially, seasonally, and with emissions scenario (see Figure 3), with the largest increases over summer and across southern England (Murphy et al., 2018). The daily maximum temperatures in summer could see even larger increases, again with the largest changes across the south. Alongside increased temperatures, it is very likely that the frequency of heatwaves and hot summer days will increase (IPCC, 2021; Murphy et al., 2018).

From a hydrological modelling perspective, changes in potential evaporation (PE) are more relevant than changes in temperature. PE is not provided as part of the UKCP18 projections, but a range of studies have calculated PE using gridded meteorological outputs from the UKCP18 RCM ensemble (e.g. Hannaford et al., 2022; Lane et al., 2022), and a UKCP18-driven PE dataset has recently been made available (Robinson et al., 2022, 2021). There are multiple different methods to calculate PE, ranging from simple temperature-based approximations to physically-based Penman-Monteith approaches, and PE estimation choices will impact the projected changes (Kay et al., 2018; Kay and Davies, 2008; Prudhomme and Williamson, 2013; Robinson et al., 2022; Rudd and Kay, 2016). PE estimates from the UKCP18 RCM ensemble using a Penman-Monteith approach have been able to capture the broad regional and seasonal patterns of observed PE across Great Britain, with higher values in the warmer and drier south-east, and higher values in the summer (Robinson et al., 2022). Robinson et al. (2022) found PE increased by around 14-29% by 2060-2080, with the largest increases in summer (18-36%) and only moderate increases (or even decreases for some ensemble members) in winter. Lane et al. (2022) found that the largest increases to PE were generally in the south-east, further amplifying the regional and seasonal differences in PE across GB, although the magnitude and spatial pattern of PE change differed between ensemble members (see Figure 4).

For precipitation, projections generally indicate decreasing summer rainfall and increasing winter rainfall for the UK (see Figure 3), but changes are highly spatially variable and dependent on climate model resolution and emissions scenario (Guillod et al., 2018; Met Office, 2019). The weather@home projections show small increases in precipitation in winter but large decreases in summer leading to an overall drying (Guillod et al., 2018; Rudd et al., 2019). The UKCP18 probabilistic projections indicate likely reductions in summer precipitation (-45% to +5%) and increases in winter precipitation (-3% to +39%) by 2070 under RCP8.5, with an uncertain sign of change for annual precipitation (Met Office, 2019; Murphy et al., 2018).

Higher-resolution simulations are better able to capture the spatial patterns and seasonal cycle of mean and extreme precipitation (Gutiérrez et al., 2021; Kendon et al., 2014). The

UKCP18 Local (2.2km) projections suggest that alongside a general summer drying trend, there could be an increase in the intensity of heavy summer downpours, and significant increases in hourly precipitation extremes in the future (Met Office, 2019). A change in the seasonality of extremes is also projected, with convective storms extending from summer into autumn (Met Office, 2019).

The impacts of these climatic changes for streamflow droughts are complex. On the one hand, an increase in temperatures (and subsequently PE) and reduction in summer rainfall could lead to reduced river flow in the future. However, these could be offset by increased winter precipitation, summer downpours and high intensity rainfall events. Charlton and Arnell (2014) find that while changes in high flows are largely driven by precipitation, changes in indicators of low flow are sensitive to changes in both summer precipitation and temperature. The river flow response to changing climate will also be strongly dependent on catchment characteristics and catchment water balance (Charlton and Arnell, 2014; Prudhomme et al., 2015). Spatial variation in climatic changes and catchment characteristics adds further complexity to projections of drought impacts at the national scale, thus requiring a hydrological modelling approach to assess changes to low flows/hydrological droughts.

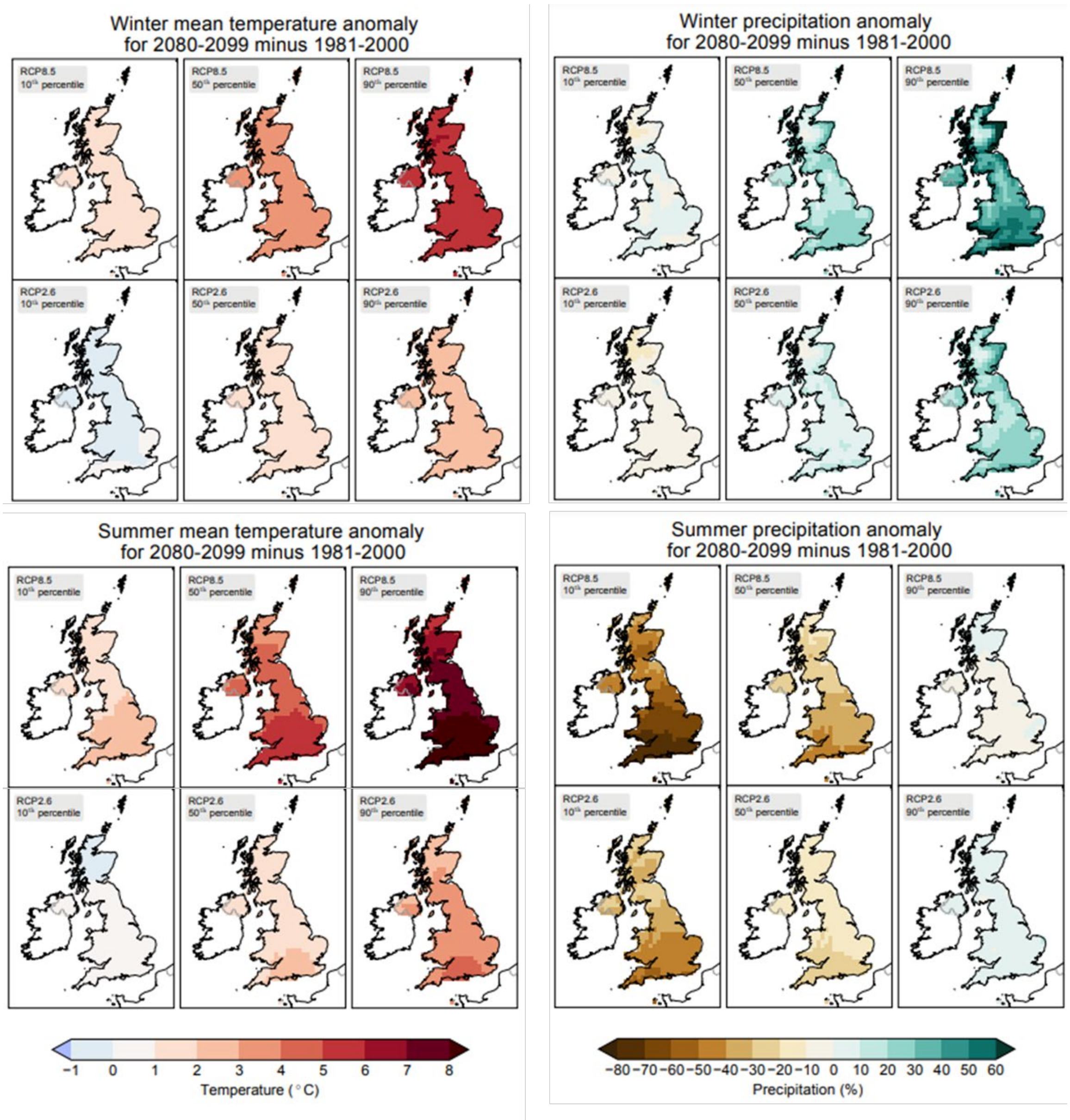


Figure 3. Projected climate changes from the UKCP18 probabilistic projections, adapted from Murphy et al. (2018). Left: changes in 20-year mean winter (top) and summer (bottom) temperature for 2 emissions scenarios (RCP8.5 above, RCP2.6 below). Columns show results for the 10th (left), 50th (middle) and 90th (right) percentile outcomes. Right: changes in the 20-year seasonal mean precipitation over winter (top) and summer (bottom).

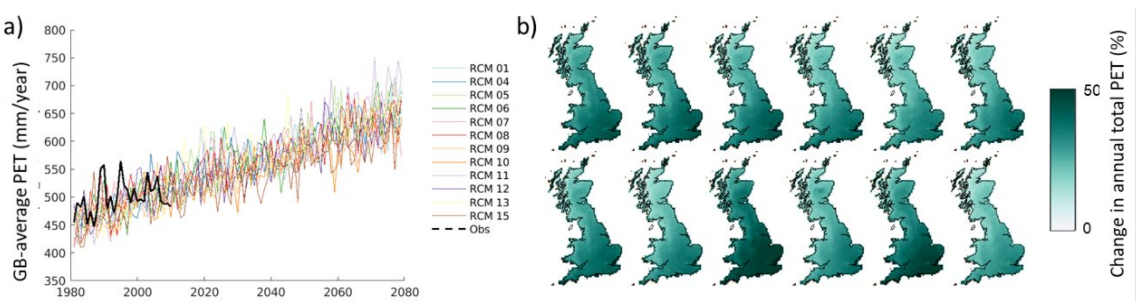
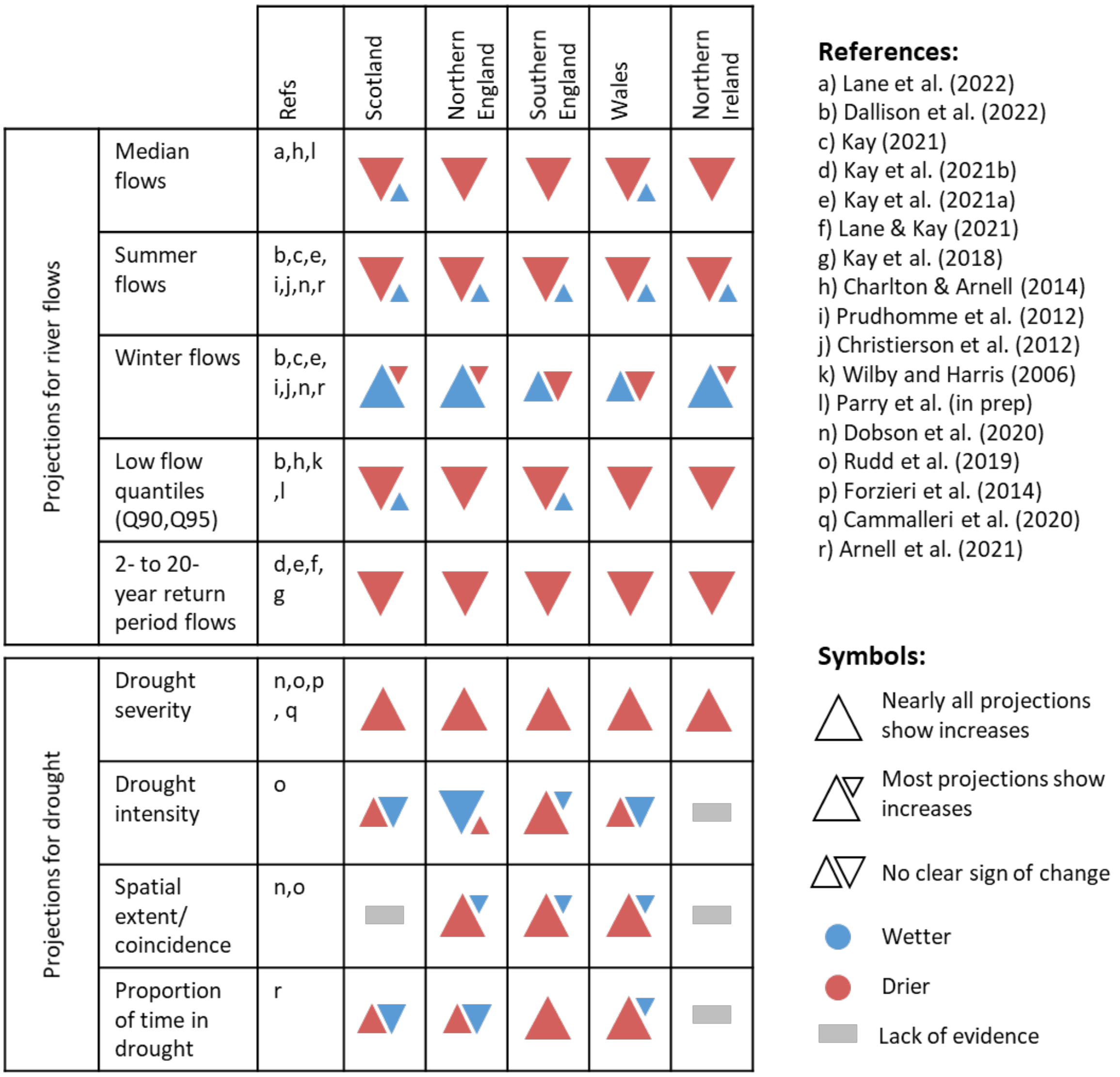


Figure 4. Projected changes in PE derived from the UKCP18 perturbed-parameter RCM ensemble (all based on RCP8.5), adapted from Lane et al. (2022). PE was estimated using a Penman-Monteith equation parameterised for short grass, and bias-corrected using the quantile-quantile method. a) Time-series of GB-average PE from 1980 to 2080, for each RCM (coloured lines), and CHESS-PE observations (black line) (Robinson et al., 2015). b) Percentage change in annual total PE between baseline (1985-2010) and future (2050-2075) periods for each RCM.

3.2. Low flow impacts

There are now many studies evaluating climate change impacts for river low flows across Great Britain (including Charlton and Arnell, 2014; Christierson et al., 2012; Kay et al., 2021b, 2018; Lane and Kay, 2021; Prudhomme et al., 2012). These use a range of different climate projections, bias-correction methods, downscaling approaches and hydrological models (see Table 1), all contributing to large ranges in projected flow impacts. Direct comparisons between studies are hampered by varying spatial coverage (e.g. selection of a different set of catchments, or gridded vs catchment-based modelling), selection of different emissions scenarios and/or time horizons, and a focus on different flow metrics. Table 2 provides a simple summary of results from the literature, focusing on the direction of change in median and low-flow metrics across the UK.

Table 2. Summary of the climate change impacts for low flows and droughts by the mid- to far-future (2050s onwards), focusing on the direction of change. Upwards pointing triangles indicate an increasing trend, and downwards pointing triangles indicate a decreasing trend for each metric. Red indicates less water in the future (i.e. lower low flows/worsening droughts) while blue indicates more water (i.e. increasing low flow magnitudes/ easing of droughts). It is important to note that this is a simplified overview of results from the literature, and there may be differences in methods, metrics, climate scenarios, hydrological models, time periods, and number of studies between regions. (Arnell et al., 2021b; Charlton and Arnell, 2014; Christierson et al., 2012; Dallison et al., 2022; Dobson et al., 2020; Forzieri et al., 2014; Kay, 2021; Kay et al., 2021b, 2021a, 2018; Lane et al., 2022; Lane and Kay, 2021; Prudhomme et al., 2012; Rudd et al., 2019; Wilby and Harris, 2006)



Projected changes in low flow magnitude

Projections generally indicate reductions to median (Q50) and summer flows, and an uncertain sign of change for winter flows (Table 2). Lane et al., (2022) evaluated changes to Q50 across 346 catchments in Great Britain, producing 360 spatially-coherent future flow scenarios comprising the 12 regional UKCP18 projections and 30 different parameterisations of the DECIPHeR hydrological model. They found reductions in Q50 for nearly all scenarios by 2050-2075, with ensemble-median changes ranging from -50% to +1% across regions. Reductions to Q50 were smallest in west Scotland, which was the only region to show possible increases within the ensemble spread. Christierson et al. (2012) evaluated changes to annual and monthly river flows using an uncertainty framework, which included the UKCP09 probabilistic projections and two hydrological models. Central estimates showed large reductions in summer flows for the period 2011-2040 (up to around -40%) and a geographic divide for winter flows, with small increases along the west coast and for NI, and decreases in south-east England. Similar patterns were found in a more recent analysis using the UKCP18 RCM ensemble, as shown in Figure 5 (Kay, 2021).

Projections for low flow quantiles (e.g. Q90, the water volume exceeded 90% of the time), also generally show reductions across the UK. For example, Charlton and Arnell (2014) find reductions in Q95 under all climate scenarios by the 2080s, when assessing changes for six British catchments using the UKCP09 probabilistic projections. Dallison et al. (2022) find reductions in low flow quantiles (Q75, Q90, Q95 and Q99) for catchments across Wales by the 2070s, with generally larger reductions for the more extreme low flow quantiles. However, Wilby and Harris (2006) demonstrate that changes in Q95 are dependent on modelling choices, particularly the choice of GCM. They look at Q95 changes for the Thames using four GCMs (CGCM2, CSIRO, ECHAM4 and HADCM3), finding that most projections indicate declining Q95, except when using the CGCM2 GCM where large projected increases in summer rainfall led to Q95 changes of -10 to +80%.

A number of studies have carried out low flow frequency analysis, to assess changes in extreme low flows (Kay et al., 2021b, 2018; Lane and Kay, 2021). These studies fit a generalised extreme value (GEV) distribution to the simulated 7-day (or 30-day) annual minimum flow, to provide low flow estimates for a given return period. Both Kay et al. (2018) and Lane and Kay (2021) found a large reduction in 2- to 20-year return period low flows across GB, with reductions in 10-year return periods of around -90% to -25% by 2050-2080 (Figure 6). Reductions were generally largest in the south, for higher return periods, and for later time periods.

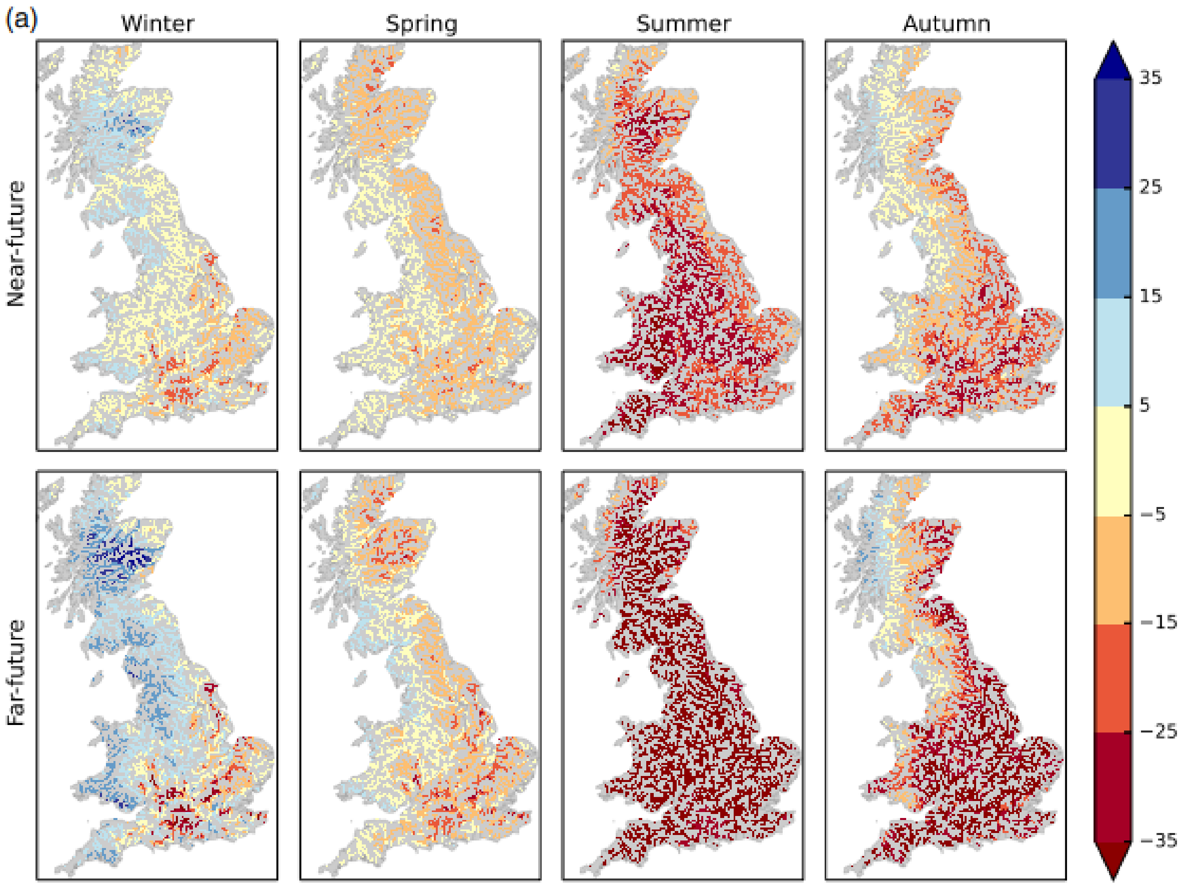


Figure 5. Percentage change in seasonal mean flow from the pooled ensemble (comprised of 12x perturbed-parameter RCM simulations from UKCP18 run through the G2G hydrological model), adapted from (Kay, 2021).

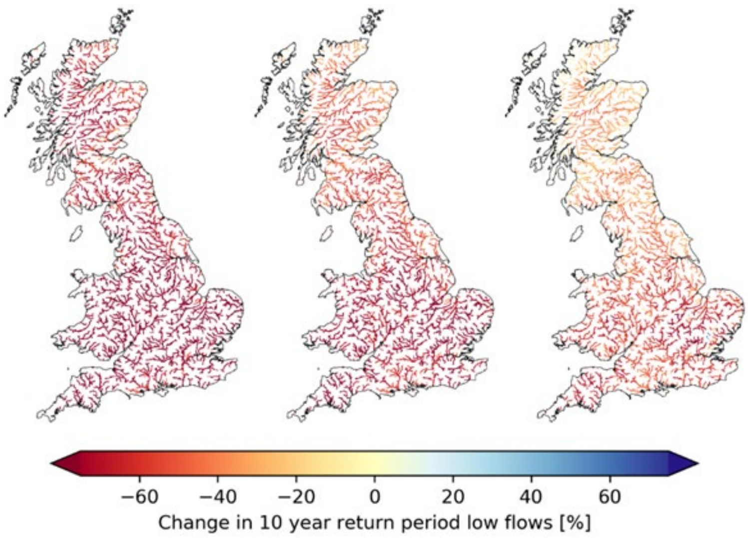


Figure 6. Percentage change in 10-year return period 7-day low flows, produced using the 12-member UKCP18 RCM ensemble and G2G hydrological model, adapted from Lane and Kay, (2021). Plots summarise the projected changes from all 12 ensemble members, giving the second lowest (left), median (middle) and second highest (right) result for each river grid cell.

Dependence on time-horizon

With most studies looking at changes in flow between fixed baseline and future periods, an interesting question is how low flows will change over time or at certain global warming levels. For instance, gradually declining low flows between the baseline and future periods may be easier to adapt to than any sudden large declines in flow if a tipping point is reached.

Many studies report decreasing low flows, with larger reductions for projections further into the future. Charlton and Arnell, (2014) find that annual runoff is more likely to decrease for later time-periods, as the effects of increased PE begin to outweigh increases in precipitation. Kay et al., (2018) also finds larger reductions in low flow extremes in later time periods. However, projections further into the future also have larger uncertainties (Wilby and Harris, 2006).

A few studies have looked at transient changes in low flow magnitude, using a moving window approach to define the future time-slices (Kay et al., 2021b). Kay et al., (2021b) find that 5-year return period 7-day low flows tend to steadily decline over time for most regions. However, there are a cluster of projections that show low flows increasing from 2030-2060 and then sharply declining from 2060-2080 in the West Highlands. There is also a cluster of projections for south-west England that show increasing low flows until 2040, after which they decline sharply. Studies using only a single future period could therefore produce misleading projections for regions with non-linear changes through time.

Arnell et al. (2021a) looked at changes to a set of climate risk indicators, including hydrological drought, at levels of warming up to 4oC above pre-industrial levels. They found that changes to the proportion of time under drought conditions showed a generally linear increase with level of warming. However, for Scotland, most of the impacts occurred at low levels of warming, with little difference in impact between 2.5oC and 4oC warming.

Influence of catchment characteristics

Projected changes in low flows vary across the UK, because of 1) spatially varying climatic changes, 2) differences in catchment properties (e.g. geology, slope, base flow index) and 3) catchment water balance (i.e. catchment wetness) over the baseline period. Charlton and Arnell (2014) modelled changes in median (Q50) and low (Q95) flows with different combinations of model parameters and climate forcing, to help explain the variations in river flow response across 6 English catchments. Even when all catchments were driven with the same climatic changes, large differences in river flow response remained, demonstrating the importance of catchment characteristics and baseline climate conditions in moderating climate change impact on flows. In particular, catchment permeability was found to have a major effect on catchment response to climatic changes, with a slightly wider range of low flow responses in the less permeable upland catchments. Catchment storage capacity is potentially the most important catchment characteristic for hydrological drought development (Van Loon, 2015). Major catchment stores include the soil column, groundwater system, peat swamps and bogs, lakes and reservoirs. Catchments with a large storage capacity tend to have a longer hydrological memory, which slows the transformation of the drought signal from a meteorological drought to a hydrological drought (Van Loon, 2015). Storage capacity of a catchment can be related to geology, topography, soil type, land use and vegetation, with no clear dominant factor for explaining drought severity (Van Loon and Laaha, 2015).

For example, Prudhomme et al. (2015) developed response surfaces looking at the relationship between changes in temperature, precipitation and the magnitude and seasonality of low flows for two contrasting catchments. They focused on the largely impermeable, upland Mint catchment in north west England and the drier, groundwater dominated Thet catchment in eastern England. For the Mint catchment, there was a clear relationship between changes in spring-summer temperature/precipitation and summer flows, likely due to the low catchment storage and corresponding short hydrological memory of the catchment. For the Thet catchment, which had a delayed climate-to-low flow relationship, the picture was more complicated. Changes in spring rainfall had the biggest impact on summer flows, but overall the relationship between climatic changes and flow couldn't easily be captured by the 2-D response surface.

3.3. Drought impacts

Overall, projections indicate an increase in the severity of future streamflow droughts across the UK, with the largest changes for higher warming levels or further into the future, as summarised in Table 2 (Cammalleri et al., 2020; Dobson et al., 2020; Feyen and Dankers, 2009; Forzieri et al., 2014; Rudd et al., 2019). This is more robust than changes in low flows, with studies finding an increase in deficit volumes (i.e. worsening drought severity) even in regions where there is little change in minimum flows (Feyen and Dankers, 2009). Changes to streamflow drought duration are more spatially variable, but catchments in southeast England have been highlighted as potential hot-spots for future multi-year droughts, with an increase in drought duration and intensity (Brunner and Tallaksen, 2019; Rudd et al., 2019).

There are relatively few studies evaluating climate change impacts for hydrological drought nationally for the UK (Arnell et al., 2021b; Dobson et al., 2020; Rudd et al., 2019). Rudd et al., (2019) evaluated changes in drought duration, intensity and severity (duration x intensity), using the weather@home climate projections and a national-scale gridded hydrological model. They found nationwide increases in drought severity and the spatial extent of droughts into the future. The largest increases in drought severity were for regions in the southeast, where projections showed increases in both drought intensity and duration. Central regions (e.g. Northwest England, Humber, and southern Scotland) showed a decrease in peak drought intensities, despite an increase in overall drought severity. Furthermore, they found a change in the seasonality of drought events, with a shift towards the largest drought events occurring later in the summer (August – October), and fewer occurring earlier in the year (Jan – July).

Dobson et al. (2020) evaluated changes to droughts and water scarcity across England and Wales, using climate simulations from the weather@home project combined with hydrological and water resource system models. Their projections showed a worsening of extreme streamflow droughts for all 80 modelled catchments, leading to the probability of a year with water use restrictions doubling by 2050.

Arnell et al. (2021b) evaluated climate change impact on streamflow droughts using the UKCP18 global projections, applied using the delta change approach. They focused on the proportion of time river flows were in drought conditions, characterised as the time with SSI < -1.5, accumulated over 12 months. Interestingly, projections using the Hadley Centre climate model showed large increases in the proportion of time under drought conditions for all regions except West Scotland and Northwest England. However, projections underpinned by the CMIP5 models showed relatively small changes to drought conditions, with the ensemble range spanning 'no change' for most regions, and the ensemble median showing reduced time in drought conditions for northern England and Scotland. This large difference in drought projections between GCMs, all using the RCP8.5 emissions scenario, is likely due to the Hadley Centre model sampling the warmer range of possible outcomes (Lowe et al., 2019), and further demonstrates the large uncertainties in future climate projections.

4. Knowledge Gaps

Remaining knowledge gaps for modelling climate change impacts on low flows and droughts include:

Representation and communication of uncertainties

As discussed above, there are large uncertainties when modelling climate change impacts on river flows, cascading through the climate-hydrological modelling chain. A key issue is ensuring that projections reflect uncertainties, and that these are clearly communicated to end users. This is challenging for large-scale studies, where model simulations can be computationally expensive and thus inclusion of multiple uncertainty sources (e.g. emissions scenarios, GCMs, hydrological models, bias-correction and downscaling approaches) is infeasible. Given that it is not possible to represent the full range of uncertainties, more pragmatic approaches to uncertainty are needed, which include consultations from decision makers on how to best meet their needs (Smith et al., 2018).

The majority of national-scale UK climate impact assessments are based on the Hadley Centre climate model, which underpins UKCP09, UKCP18 Regional and climateprediction.net/weather@home, albeit with substantial model improvements over time. GCMs are a large source of uncertainty in climate impact studies, and over-reliance on a single GCM can lead to misleading conclusions. For example, Arnell et al. (2021b) found large increases in future drought for most regions of GB using the Hadley Centre GCM, but only small changes when basing projections on the other CMIP5 GCMs. There is therefore a need for more studies including and contrasting UK-wide drought impacts from the Hadley Centre and other GCMs within a common framework. This could be enabled by the EuroCORDEX-UK projections, which are extending the UKCP18 suite of climate projections with high-resolution climate projections from a broader range of climate models (Chandler, 2023). Hydrological modelling uncertainties may be particularly relevant for low flows and droughts, but are usually not included in national climate impact studies for drought. Developing frameworks to support the use of multiple model structures could help facilitate multi-model studies in the future.

Understanding possible low-likelihood but high-impact events

Given large uncertainties in future projections, many studies focus on the most likely changes within the ensemble, for example reporting changes in an ensemble mean. However, it is also useful to understand possible extreme events which are within the envelope of current or future natural variability. These can be used to stress-test systems and ensure adaptation measures are robust to worst-case scenarios. Large climate ensembles currently under development, for example Single-Model Initial-condition Large Ensembles (SMILEs) within the CANARI project (NCAS, 2023), could help to facilitate projections of low-likelihood but high-impact drought events. Bottom-up approaches, such as scenario-neutral response surfaces or storylines of extreme scenarios (e.g. from UK Climate Resilience Programme, 2023), could also help to better understand the plausible drivers and pathways of low-likelihood, high-impact droughts.

Understanding climate change impacts on drought duration, intensity, frequency and spatial extent at the national scale

There are many studies evaluating climate change impact on seasonal and low river flows across Great Britain, but few studies evaluating changes to streamflow drought over multiple seasons. The two recent studies to evaluate climate impacts on streamflow drought from RCM projections at a national scale (Dobson et al., 2020; Rudd et al., 2019) both use weather@home simulations, so while they capture uncertainties due to climate variability, they are based on a single GCM structure. Arnell et al. (2021b) applied multiple GCMs, but the use of a delta-change method meant that it was not possible to assess changes in drought spatial coherence or the sequencing of events. Further studies using a range of climate projection products are required to give a more complete picture of changing drought risk across the UK.

Improving hydrological modelling of low flows

Possible improvements to ensure hydrological models are suitable for simulating climate change impacts on low flows include: 1) adding/improving representation of human-water interactions, 2) calibration and evaluation focused on low flow / drought representation, 3) developing model structures/ parameterisations that are robust to a non-stationary climate, including hydrological system shifts that can occur following extreme droughts, 4) including multiple model structures/parameter sets for more robust future flow projections.

Understanding climate change impacts in the context of other changes

This review has focused on how climate change may impact future river flows and droughts. However, other factors such as land-use change and changing abstractions/discharges will lead to altered flows in the future. The development of more holistic scenarios of future changes in a range of environmental drivers could help facilitate modelling studies looking at changes in climate, population and land-cover changes within a consistent framework. For example, the SPEED project is producing spatially-explicit projections including future climate, socioeconomic pathways and land use change (UKCEH, 2023).

5. Summary

Streamflow droughts could be exacerbated by climate change, with serious consequences spanning multiple sectors. In order to develop robust adaptation measures, we need to understand how the characteristics of streamflow droughts could change in the future. This review summarised the state-of-the-art in modelling climate change impacts for low flows and streamflow drought across the UK, with a focus on remaining limitations.

Approaches to model climate change impact on UK droughts were found to be generally ‘top-down’ in nature, using a climate-hydrological modelling cascade. However, there have also been applications of ‘bottom-up’ approaches focusing on system sensitivity to change and plausible future extremes, which can complement climate model-driven studies. Ongoing methodological difficulties include: 1) representation and communication of modelling uncertainties, which can be very large for future river flow / drought projections; 2) using climate model data for hydrological applications, coping with disparities in resolution and biases in key hydrological variables; 3) development of national-scale hydrological models which can robustly simulate low flows under non-stationary climate conditions, and include other environmental or anthropogenic changes such as land-use change and changes to licensed abstractions and discharges.

Projected changes in low flows and droughts were found to vary across the UK, due to spatially varying climatic changes, differences in catchment properties such as storage capacity, and the catchment water balance over the baseline period. However, projections generally showed a reduction in median flows, summer flows and low flows, with larger reductions further into the future. Changes to winter flows were less clear, with generally increasing flows across Scotland, northern England and Northern Ireland, and a mixed picture elsewhere. For droughts, most studies found that climate change led to increased drought severity across the UK, but changes to drought intensity and duration varied regionally. Southeast England was highlighted as a hot-spot for future multi-year droughts, with projected increases in drought severity, intensity and duration.

The need for national-scale drought impact studies comparing and contrasting results from a range of climate models was highlighted as a key research gap, as most UK-wide studies are based on the Hadley Centre climate model. Other research gaps identified were: representation of hydrological modelling uncertainties in national scale climate change impact studies; developing scenarios of low-likelihood but high impact events to ensure adaptation measures are robust; developing hydrological models that can robustly simulate low flows under non-stationary conditions; and further understanding changes to drought duration, intensity, frequency, spatial extent and coincidence at the national scale.

References

Addor, N., Rössler, O., Köplin, N., Huss, M., Weingartner, R., Seibert, J., 2014. Robust changes and sources of uncertainty in the projected hydrological regimes of Swiss catchments. Water Resour. Res. 50, 7541–7562. https://doi.org/10.1002/2014WR015549

Addor, N., Seibert, J., 2014. Bias correction for hydrological impact studies - beyond the daily perspective. Hydrol. Process. 28, 4823–4828. https://doi.org/10.1002/hyp.10238

Arnell, N.W., Freeman, A., Kay, A.L., Rudd, A.C., Lowe, J.A., 2021a. Indicators of climate risk in the UK at different levels of warming. Environ. Res. Commun. 3, 095005. https://doi.org/10.1088/2515-7620/ac24c0

Arnell, N.W., Kay, A.L., Freeman, A., Rudd, A.C., Lowe, J.A., 2021b. Changing climate risk in the UK: A multi-sectoral analysis using policy-relevant indicators. Clim. Risk Manag. 31, 100265. https://doi.org/10.1016/j.crm.2020.100265

Aryal, A., Shrestha, S., Babel, M.S., 2018. Quantifying the sources of uncertainty in an ensemble of hydrological climate-impact projections. Theor. Appl. Climatol. https://doi.org/10.1007/s00704-017-2359-3

Barker, L.J., Hannaford, J., Parry, S., Smith, K.A., Tanguy, M., Prudhomme, C., 2019. Historic hydrological droughts 1891–2015: systematic characterisation for a diverse set of catchments across the UK. Hydrol. Earth Syst. Sci. 23, 4583–4602. https://doi.org/10.5194/hess-23-4583-2019

Bastola, S., Murphy, C., Sweeney, J., 2011. The role of hydrological modelling uncertainties in climate change impact assessments of Irish river catchments. Adv. Water Resour. 34, 562–576. https://doi.org/10.1016/j.advwatres.2011.01.008

Bell, V.A., Kay, A.L., Cole, S.J., Jones, R.G., Moore, R.J., Reynard, N.S., 2012. How might climate change affect river flows across the Thames Basin? An area-wide analysis using the UKCP09 Regional Climate Model ensemble. J. Hydrol. 442–443, 89–104. https://doi.org/10.1016/j.jhydrol.2012.04.001

Bell, V.A., Kay, A.L., Jones, R.G., Moore, R.J., Reynard, N.S., 2009. Use of soil data in a grid-based hydrological model to estimate spatial variation in changing flood risk across the UK. J. Hydrol. 377, 335–350. https://doi.org/10.1016/j.jhydrol.2009.08.031

Bergström, S., 1992. The HBV model - its structure and applications. Swedish Meteorol. Hydrol. Institute, Norrköping.

Bloomfield, J., Jackson, C., 2023. Groundwater droughts in England: a review of current understanding and knowledge gaps, and opportunities and priorities for future research.

Brigode, P., Oudin, L., Perrin, C., 2013. Hydrological model parameter instability: A source of additional uncertainty in estimating the hydrological impacts of climate change? J. Hydrol. 476, 410–425. https://doi.org/10.1016/j.jhydrol.2012.11.012

Brunner, M.I., Tallaksen, L.M., 2019. Proneness of European Catchments to Multiyear Streamflow Droughts. Water Resour. Res. 55, 8881–8894. https://doi.org/10.1029/2019WR025903

Cammalleri, C., Naumann, G., Mentaschi, L., Bisselink, B., Gelati, E., De Roo, A., Feyen, L., 2020. Diverging hydrological drought traits over Europe with global warming. Hydrol. Earth Syst. Sci. 24, 5919–5935. https://doi.org/10.5194/hess-24-5919-2020

Chan, W.C.H., Shepherd, T.G., Facer-Childs, K., Darch, G., Arnell, N.W., 2022a. Tracking the methodological evolution of climate change projections for UK river flows. Prog. Phys. Geogr. Earth Environ. 46, 589–612. https://doi.org/10.1177/03091333221079201

Chan, W.C.H., Shepherd, T.G., Facer-Childs, K., Darch, G., Arnell, N.W., 2022b. Storylines of UK drought based on the 2010–2012 event. Hydrol. Earth Syst. Sci. 26, 1755–1777. https://doi.org/10.5194/hess-26-1755-2022

Chandler, R., 2023. EuroCORDEX-UK: Enabling the use and producing improved understanding of EuroCORDEX data over the UK [WWW Document]. URL https://www.ucl.ac.uk/statistics/research/eurocordex-uk (accessed 1.20.23).

Charlton, M.B., Arnell, N.W., 2014. Assessing the impacts of climate change on river flows in England using the UKCP09 climate change projections. J. Hydrol. 519, 1723–1738. https://doi.org/10.1016/j.jhydrol.2014.09.008

Chegwidden, O.S., Nijssen, B., Rupp, D.E., Arnold, J.R., Clark, M.P., Hamman, J.J., Kao, S., Mao, Y., Mizukami, N., Mote, P.W., Pan, M., Pytlak, E., Xiao, M., 2019. How Do Modeling Decisions Affect the Spread Among Hydrologic Climate Change Projections? Exploring a Large Ensemble of Simulations Across a Diversity of Hydroclimates. Earth's Futur. 7, 623–637. https://doi.org/10.1029/2018EF001047

Christierson, B. v., Vidal, J.-P., Wade, S.D., 2012. Using UKCP09 probabilistic climate information for UK water resource planning. J. Hydrol. 424–425, 48–67. https://doi.org/10.1016/j.jhydrol.2011.12.020

Clark, M.P., Slater, A.G., Rupp, D.E., Woods, R.A., Vrugt, J.A., Gupta, H. V., Wagener, T., Hay, L.E., 2008. Framework for Understanding Structural Errors (FUSE): A modular framework to diagnose differences between hydrological models. Water Resour. Res. 44, 1–14. https://doi.org/10.1029/2007WR006735

Cloke, H.L., Jeffers, C., Wetterhall, F., Byrne, T., Lowe, J., Pappenberger, F., 2010. Climate impacts on river flow: projections for the Medway catchment, UK, with UKCP09 and CATCHMOD. Hydrol. Process. 24, 3476–3489. https://doi.org/10.1002/hyp.7769

Coron, L., Andréassian, V., Perrin, C., Lerat, J., Vaze, J., Bourqui, M., Hendrickx, F., 2012. Crash testing hydrological models in contrasted climate conditions: An experiment on 216 Australian catchments. Water Resour. Res. 48, 1–17. https://doi.org/10.1029/2011WR011721

Counsel, C., Durant, M., 2023. Review of the state of research on drought - what we do and do not know about drought - water supply; observed and projected.

Coxon, G., Freer, J., Lane, R.A., Dunne, T., Knoben, W.J.M., Howden, N.J.K., Quinn, N., Wagener, T., Woods, R., 2019. DECIPHeR v1: Dynamic fluxEs and ConnectIvity for Predictions of HydRology. Geosci. Model Dev. 12, 2285–2306. https://doi.org/10.5194/gmd-12-2285-2019

Dallison, R.J.H., Williams, A.P., Harris, I.M., Patil, S.D., 2022. Modelling the impact of future climate change on streamflow and water quality in Wales, UK. Hydrol. Sci. J. 67, 939–962. https://doi.org/10.1080/02626667.2022.2044045

De Niel, J., Van Uytven, E., Willems, P., 2019. Uncertainty Analysis of Climate Change Impact on River Flow Extremes Based on a Large Multi-Model Ensemble. Water Resour. Manag. 33, 4319–4333. https://doi.org/10.1007/s11269-019-02370-0

Dobson, B., Coxon, G., Freer, J., Gavin, H., Mortazavi-Naeini, M., Hall, J.W., 2020. The Spatial Dynamics of Droughts and Water Scarcity in England and Wales. Water Resour. Res. 56. https://doi.org/10.1029/2020WR027187

Ehret, U., Zehe, E., Wulfmeyer, V., Warrach-Sagi, K., Liebert, J., 2012. HESS Opinions “should we apply bias correction to global and regional climate model data?” Hydrol. Earth Syst. Sci. 16, 3391–3404. https://doi.org/10.5194/hess-16-3391-2012

Feyen, L., Dankers, R., 2009. Impact of global warming on streamflow drought in Europe. J. Geophys. Res. 114, D17116. https://doi.org/10.1029/2008JD011438

Forzieri, G., Feyen, L., Rojas, R., Flörke, M., Wimmer, F., Bianchi, A., 2014. Ensemble projections of future streamflow droughts in Europe. Hydrol. Earth Syst. Sci. 18, 85–108. https://doi.org/10.5194/hess-18-85-2014

Fowler, K., Peel, M., Saft, M., Nathan, R., Horne, A., Wilby, R., McCutcheon, C., Peterson, T., 2022. Hydrological Shifts Threaten Water Resources. Water Resour. Res. 58. https://doi.org/10.1029/2021WR031210

Guillod, B.P., Jones, R.G., Dadson, S.J., Coxon, G., Bussi, G., Freer, J., Kay, A.L., Massey, N.R., Sparrow, S.N., Wallom, D.C.H., Allen, M.R., Hall, J.W., 2018. A large set of potential past, present and future hydro-meteorological time series for the UK. Hydrol. Earth Syst. Sci. 22, 611–634. https://doi.org/10.5194/hess-22-611-2018

Gutiérrez, J.M., Jones, R.G., Narisma, G.T., Alves, L.M., Amjad, M., Gorodetskaya, I.V., Grose, M., Klutse, N.A.B., Krakovska, S., Li, J., Martínez-Castro, D., Mearns, L.O., Mernild, S.H., Ngo-Duc, T., Hurk, B. van den, Yoon, J.-H., 2021. Atlas, in: Masson-Delmotte, V., Zhai, P., Pirani, A., Connors, S.L., Péan, C., Berger, S., Caud, N., Chen, Y., Goldfarb, L., Gomis, M.I., Huang, M., Leitzell, K., Lonnoy, E., Matthews, J.B.R., Maycock, T.K., Waterfield, T., Yelekçi, O., Yu, R., Zhou, B. (Eds.), Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 1927–2058. https://doi.org/10.1017/9781009157896.021

Hagemann, S., Chen, C., Clark, D.B., Folwell, S., Gosling, S.N., Haddeland, I., Hanasaki, N., Heinke, J., Ludwig, F., Voss, F., Wiltshire, A.J., 2013. Climate change impact on available water resources obtained using multiple global climate and hydrology models. Earth Syst. Dyn. 4, 129–144. https://doi.org/10.5194/esd-4-129-2013

Hallouin, T., Ellis, R.J., Clark, D.B., Dadson, S.J., Hughes, A.G., Lister, B.N.L.G.M.S., Polcher, J., 2021. UniFHy v0.1: A community framework for the terrestrial water cycle in Python. Geosci. Model Dev. Discuss. https://doi.org/10.5194/gmd-2021-419

Hannaford, J., Barker, L., Turner, S., Parry, S., Tanguy, M., Chevuturi, A., 2023. Past variability in observed hydrological (river flow) drought.

Hannaford, J., Mackay, J.D., Ascott, M., Victoria, A., Chitson, T., Cole, S., Counsell, C., Durant, M., Jackson, C.R., Kay, A.L., Lane, R.A., Moore, R., Parry, S., Rudd, A.C., Facer-childs, K., Turner, S., Wallbank, J.R., 2022. eFLaG: enhanced future FLows and Groundwater. A national dataset of hydrological projections based on UKCP18. Earth Syst. Sci. Data Discuss. https://doi.org/10.5194/essd-2022-40

Horton, P., Schaefli, B., Kauzlaric, M., 2022. Why do we have so many different hydrological models? A review based on the case of Switzerland. WIREs Water 9. https://doi.org/10.1002/wat2.1574

IPCC, 2021. Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge Univeristy Press, Cambridge, United Kingdom and New York. https://doi.org/10.1017/9781009157896

Karlsson, I.B., Sonnenborg, T.O., Refsgaard, J.C., Trolle, D., Børgesen, C.D., Olesen, J.E., Jeppesen, E., Jensen, K.H., 2016. Combined effects of climate models, hydrological model structures and land use scenarios on hydrological impacts of climate change. J. Hydrol. 535, 301–317. https://doi.org/10.1016/j.jhydrol.2016.01.069

Kay, A.L., 2022. Differences in hydrological impacts using regional climate model and nested convection-permitting model data. Clim. Change 173, 11. https://doi.org/10.1007/s10584-022-03405-z

Kay, A.L., 2021. Simulation of river flow in Britain under climate change: Baseline performance and future seasonal changes. Hydrol. Process. 35. https://doi.org/10.1002/hyp.14137

Kay, A.L., Bell, V.A., Guillod, B.P., Jones, R.G., Rudd, A.C., 2018. National-scale analysis of low flow frequency: historical trends and potential future changes. Clim. Change 147, 585–599. https://doi.org/10.1007/s10584-018-2145-y

Kay, A.L., Crooks, S.M., Reynard, N.S., 2014. Using response surfaces to estimate impacts of climate change on flood peaks: Assessment of uncertainty. Hydrol. Process. 28, 5273–5287. https://doi.org/10.1002/hyp.10000

Kay, A.L., Davies, H.N., 2008. Calculating potential evaporation from climate model data: A source of uncertainty for hydrological climate change impacts. J. Hydrol. https://doi.org/10.1016/j.jhydrol.2008.06.005

Kay, A.L., Davies, H.N., Bell, V.A., Jones, R.G., 2009. Comparison of uncertainty sources for climate change impacts: flood frequency in England. Clim. Change 92, 41–63. https://doi.org/10.1007/s10584-008-9471-4

Kay, A.L., Davies, H.N., Lane, R.A., Rudd, A.C., Bell, V.A., 2021a. Grid-based simulation of river flows in Northern Ireland: Model performance and future flow changes. J. Hydrol. Reg. Stud. 38, 100967. https://doi.org/10.1016/j.ejrh.2021.100967

Kay, A.L., Griffin, A., Rudd, A.C., Chapman, R.M., Bell, V.A., Arnell, N.W., 2021b. Climate change effects on indicators of high and low river flow across Great Britain. Adv. Water Resour. 151, 103909. https://doi.org/10.1016/j.advwatres.2021.103909

Kay, A.L., Jones, R.G., 2012. Comparison of the use of alternative UKCP09 products for modelling the impacts of climate change on flood frequency. Clim. Change 114, 211–230. https://doi.org/10.1007/s10584-011-0395-z

Kay, A.L., Rudd, A.C., Coulson, J., 2023. Spatial downscaling of precipitation for hydrological modelling: Assessing a simple method and its application under climate change in Britain. Hydrol. Process. 37. https://doi.org/10.1002/hyp.14823

Kay, A.L., Rudd, A.C., Fry, M., Nash, G., Allen, S., 2021c. Climate change impacts on peak river flows: Combining national-scale hydrological modelling and probabilistic projections. Clim. Risk Manag. 31, 100263. https://doi.org/10.1016/j.crm.2020.100263

Kay, A.L., Watts, G., Wells, S.C., Allen, S., 2020. The impact of climate change on U. K. river flows: A preliminary comparison of two generations of probabilistic climate projections. Hydrol. Process. 34, 1081–1088. https://doi.org/10.1002/hyp.13644

Kendon, E., Fosser, G., Murphy, J., Chan, S., Clark, R., Harris, G., Lock, A., Lowe, J., Martin, G., Pirret, J., Roberts, N., Sanderson, M., Tucker, S., 2019. UKCP Convection-permitting model projections: Science report.

Kendon, E.J., Roberts, N.M., Fowler, H.J., Roberts, M.J., Chan, S.C., Senior, C.A., 2014. Heavier summer downpours with climate change revealed by weather forecast resolution model. Nat. Clim. Chang. 4. https://doi.org/10.1038/NCLIMATE2258

Klemes, V., 1986. Operational testing of hydrological simulation models. Hydrol. Sci. J. 31, 13–24. https://doi.org/10.1080/02626668609491024

Knoben, W.J.M., Freer, J.E., Fowler, K.J.A., Peel, M.C., Woods, R.A., 2019. Modular Assessment of Rainfall-Runoff Models Toolbox (MARRMoT) v1.2: An open-source, extendable framework providing implementations of 46 conceptual hydrologic models as continuous state-space formulations. Geosci. Model Dev. https://doi.org/10.5194/gmd-12-2463-2019

Lane, R.A., Coxon, G., Freer, J., Seibert, J., Wagener, T., 2022. A large-sample investigation into uncertain climate change impacts on high flows across Great Britain. Hydrol. Earth Syst. Sci. 26, 5535–5554. https://doi.org/10.5194/hess-26-5535-2022

Lane, R.A., Coxon, G., Freer, J.E., Wagener, T., Johnes, P.J., Bloomfield, J.P., Greene, S., Macleod, C.J.A., Reaney, S.M., 2019. Benchmarking the predictive capability of hydrological models for river flow and flood peak predictions across over 1000 catchments in Great Britain. Hydrol. Earth Syst. Sci. 23, 4011–4032. https://doi.org/10.5194/hess-23-4011-2019

Lane, R.A., Freer, J.E., Coxon, G., Wagener, T., 2021. Incorporating Uncertainty Into Multiscale Parameter Regionalization to Evaluate the Performance of Nationally Consistent Parameter Fields for a Hydrological Model. Water Resour. Res. 57. https://doi.org/10.1029/2020WR028393

Lane, R.A., Kay, A.L., 2021. Climate Change Impact on the Magnitude and Timing of Hydrological Extremes Across Great Britain. Front. Water 3. https://doi.org/10.3389/frwa.2021.684982

Lopez, A., Fung, F., New, M., Watts, G., Weston, A., Wilby, R.L., 2009. From climate model ensembles to climate change impacts and adaptation: A case study of water resource management in the southwest of England. Water Resour. Res. 45. https://doi.org/10.1029/2008WR007499

Lowe, J.A., Bernie, D., Bett, P., Bricheno, L., Brown, S., Calvert, D., Clark, R., Eagle, K., Edwards, T., Fosser, G., Fung, F., Gohar, L., Good, P., Gregory, J., Harris, G., Howard, T., Kaye, N., Kendon, E., Krijnen, J., Maisey, P., McDonald, R., McInnes, R., McSweeney, C., Mitchell, J., Murphy, J., Palmer, M., Roberts, C., Rostron, J., Sexton, D., Thornton, H., Tinker, J., Tucker, S., Yamazaki, K., Belcher, S., 2019. UKCP18 Science Overview Report: Version 2.0.

Maraun, D., Shepherd, T.G., Widmann, M., Zappa, G., Walton, D., Gutiérrez, J.M., Hagemann, S., Richter, I., Soares, P.M.M., Hall, A., Mearns, L.O., 2017. Towards process-informed bias correction of climate change simulations. Nat. Clim. Chang. 7, 764–773. https://doi.org/10.1038/nclimate3418

Marx, A., Kumar, R., Thober, S., Rakovec, O., Wanders, N., Zink, M., Wood, E.F., Pan, M., Sheffield, J., Samaniego, L., 2018. Climate change alters low flows in Europe under global warming of 1.5, 2, and 3 °C. Hydrol. Earth Syst. Sci. 22, 1017–1032. https://doi.org/10.5194/hess-22-1017-2018

Mendoza, P.A., Clark, M.P., Mizukami, N., Gutmann, E.D., Arnold, J.R., Brekke, L.D., Rajagopalan, B., 2016. How do hydrologic modeling decisions affect the portrayal of climate change impacts? Hydrol. Process. 30, 1071–1095. https://doi.org/10.1002/hyp.10684

Mendoza, P.A., Clark, M.P., Mizukami, N., Newman, A.J., Barlage, M., Gutmann, E.D., Rasmussen, R.M., Rajagopalan, B., Brekke, L.D., Arnold, J.R., 2015. Effects of Hydrologic Model Choice and Calibration on the Portrayal of Climate Change Impacts. J. Hydrometeorol. 16, 762–780. https://doi.org/10.1175/JHM-D-14-0104.1

Meresa, H.K., Romanowicz, R.J., 2017. The critical role of uncertainty in projections of hydrological extremes. Hydrol. Earth Syst. Sci. 21, 4245–4258. https://doi.org/10.5194/hess-21-4245-2017

Met Office, 2019. UK Climate Projections: Headline Findings. Exeter.

Mizukami, N., Clark, M.P., Newman, A.J., Wood, A.W., Gutmann, E.D., Nijssen, B., Rakovec, O., Samaniego, L., 2017. Towards seamless large-domain parameter estimation for hydrologic models. Water Resour. Res. 53, 8020–8040. https://doi.org/10.1002/2017WR020401

Muerth, M.J., Gauvin St-Denis, B., Ricard, S., Velázquez, J.A., Schmid, J., Minville, M., Caya, D., Chaumont, D., Ludwig, R., Turcotte, R., 2013. On the need for bias correction in regional climate scenarios to assess climate change impacts on river runoff. Hydrol. Earth Syst. Sci. 17, 1189–1204. https://doi.org/10.5194/hess-17-1189-2013

Murphy, J.M., Harris, G.R., Sexton, D.M.H., Kendon, E.J., Bett, P.E., Clark, R.T., Eagle, K.E., Fosser, G., Fung, F., Lowe, J.A., McDonald, R.E., McInnes, R.N., McSweeney, C.F., Mitchell, J.F.B., Rostron, J.W., Thornton, H.E., Tucker, S., Yamazaki, K., Murphy, 2018. UKCP18 Land Projections: Science Report, UKCP18 Land Projections: Science Report.

NCAS, 2023. Climate change in the Arctic – North Atlantic region and impacts on the UK [WWW Document]. URL https://ncas.ac.uk/our-science/long-term-collaborations/canari/ (accessed 1.20.23).

New, M., Lopez, A., Dessai, S., Wilby, R., 2007. Challenges in using probabilistic climate change information for impact assessments: an example from the water sector. Philos. Trans. R. Soc. A Math. Phys. Eng. Sci. 365, 2117–2131. https://doi.org/10.1098/rsta.2007.2080

Parry, S., Mackay, J., Chitson, T., Hannaford, J., Bell, V.A., Lane, R.A., Moore, R., Turner, S., 2023. Divergent future drought projections in UK river flows and groundwater levels. Hydrol. Earth Syst. Sci. Discuss. [preprint] in review. https://doi.org/10.5194/hess-2023-59

Perrin, C., Michel, C., Andréassian, V., 2003. Improvement of a parsimonious model for streamflow simulation. J. Hydrol. https://doi.org/10.1016/S0022-1694(03)00225-7

Prudhomme, C., Kay, A.L., Crooks, S., Reynard, N., 2013. Climate change and river flooding: Part 2 sensitivity characterisation for british catchments and example vulnerability assessments. Clim. Change 119, 949–964. https://doi.org/10.1007/s10584-013-0726-3

Prudhomme, C., Sauquet, E., Watts, G., 2015. Low Flow Response Surfaces for Drought Decision Support: A Case Study from the UK. J. Extrem. Events 02, 1550005. https://doi.org/10.1142/S2345737615500050

Prudhomme, C., Wilby, R.L., Crooks, S., Kay, A.L., Reynard, N.S., 2010. Scenario-neutral approach to climate change impact studies: Application to flood risk. J. Hydrol. 390, 198–209. https://doi.org/10.1016/j.jhydrol.2010.06.043

Prudhomme, C., Williamson, J., 2013. Derivation of RCM-driven potential evapotranspiration for hydrological climate change impact analysis in Great Britain: A comparison of methods and associated uncertainty in future projections. Hydrol. Earth Syst. Sci. https://doi.org/10.5194/hess-17-1365-2013

Prudhomme, C., Young, A., Watts, G., Haxton, T., Crooks, S., Williamson, J., Davies, H., Dadson, S., Allen, S., 2012. The drying up of Britain? A national estimate of changes in seasonal river flows from 11 Regional Climate Model simulations. Hydrol. Process. 26, 1115–1118. https://doi.org/10.1002/hyp.8434

Quinn, J.D., Hadjimichael, A., Reed, P.M., Steinschneider, S., 2020. Can Exploratory Modeling of Water Scarcity Vulnerabilities and Robustness Be Scenario Neutral? Earth's Futur. 8. https://doi.org/10.1029/2020EF001650

Rahiz, M., New, M., 2013. 21st Century Drought Scenarios for the UK. Water Resour. Manag. 27, 1039–1061. https://doi.org/10.1007/s11269-012-0183-1

Rameshwaran, P., Bell, V.A., Brown, M.J., Davies, H.N., Kay, A.L., Rudd, A.C., Sefton, C., 2022. Use of Abstraction and Discharge Data to Improve the Performance of a National-Scale Hydrological Model. Water Resour. Res. 58. https://doi.org/10.1029/2021WR029787

Robinson, E., Blyth, E., Clark, D., Finch, J., Rudd, A., 2015. Climate hydrology and ecology research support system potential evapotranspiration dataset for Great Britain (1961-2012) [CHESS-PE]. https://doi.org/10.5285/d329f4d6-95ba-4134-b77a-a377e0755653

Robinson, E.L., Brown, M.J., Kay, A.L., Lane, R.A., Chapman, R., Bell, V.A., Blyth, E.M., 2022. Hydro-PE: gridded datasets of historical and future Penman-Monteith potential evaporation for the United Kingdom. Earth Syst. Sci. Data Discuss. [preprint] in review. https://doi.org/10.5194/essd-2022-288

Robinson, E.L., Kay, A.L., Brown, M.J., Chapman, R.M., Bell, V.A., Blyth, E.M., 2021. Potential evapotranspiration derived from the UK Climate Projections 2018 Regional Climate Model ensemble 1980-2080 (Hydro-PE UKCP18 RCM). https://doi.org/10.5285/eb5d9dc4-13bb-44c7-9bf8-c5980fcf52a4

Rudd, A.C., Kay, A.L., 2016. Use of very high resolution climate model data for hydrological modelling: estimation of potential evaporation. Hydrol. Res. 47, 660–670. https://doi.org/10.2166/nh.2015.028

Rudd, A.C., Kay, A.L., Bell, V.A., 2019. National-scale analysis of future river flow and soil moisture droughts: potential changes in drought characteristics. Clim. Change 156, 323–340. https://doi.org/10.1007/s10584-019-02528-0

Samaniego, L., Kumar, R., Attinger, S., 2010. Multiscale parameter regionalization of a grid-based hydrologic model at the mesoscale. Water Resour. Res. 46, 1–25. https://doi.org/10.1029/2008WR007327

Samaniego, L., Kumar, R., Thober, S., Rakovec, O., Zink, M., Wanders, N., Eisner, S., Müller Schmied, H., Sutanudjaja, E., Warrach-Sagi, K., Attinger, S., 2017. Toward seamless hydrologic predictions across spatial scales. Hydrol. Earth Syst. Sci. 21, 4323–4346. https://doi.org/10.5194/hess-21-4323-2017

Seiller, G., Anctil, F., Perrin, C., 2012. Multimodel evaluation of twenty lumped hydrological models under contrasted climate conditions. Hydrol. Earth Syst. Sci. https://doi.org/10.5194/hess-16-1171-2012

Slater, L.J., Anderson, B., Buechel, M., Dadson, S., Han, S., Harrigan, S., Kelder, T., Kowal, K., Lees, T., Matthews, T., Murphy, C., Wilby, R.L., 2021. Nonstationary weather and water extremes: a review of methods for their detection, attribution, and management. Hydrol. Earth Syst. Sci. 25, 3897–3935. https://doi.org/10.5194/hess-25-3897-2021

Smith, A., Freer, J., Bates, P., Sampson, C., 2014. Comparing ensemble projections of flooding against flood estimation by continuous simulation. J. Hydrol. 511, 205–219. https://doi.org/10.1016/j.jhydrol.2014.01.045

Smith, K.A., Barker, L.J., Tanguy, M., Parry, S., Harrigan, S., Legg, T.P., Prudhomme, C., Hannaford, J., 2019. A multi-objective ensemble approach to hydrological modelling in the UK: an application to historic drought reconstruction. Hydrol. Earth Syst. Sci. 23, 3247–3268. https://doi.org/10.5194/hess-23-3247-2019

Smith, K.A., Wilby, R.L., Broderick, C., Prudhomme, C., Matthews, T., Harrigan, S., Murphy, C., 2018. Navigating Cascades of Uncertainty — As Easy as ABC? Not Quite…. J. Extrem. Events 05, 1850007. https://doi.org/10.1142/S2345737618500070

Steele-Dunne, S., Lynch, P., McGrath, R., Semmler, T., Wang, S., Hanafin, J., Nolan, P., 2008. The impacts of climate change on hydrology in Ireland. J. Hydrol. https://doi.org/10.1016/j.jhydrol.2008.03.025

Tanguy, M., Chevuturi, A., Marchant, B., Mackay, J., Parry, S., Hannaford, J., n.d. How will climate change affect spatial coherence of streamflow and groundwater droughts. Submitted to Environ. Res. Lett..

Tian, Y., Xu, Y.P., Zhang, X.J., 2013. Assessment of Climate Change Impacts on River High Flows through Comparative Use of GR4J, HBV and Xinanjiang Models. Water Resour. Manag. https://doi.org/10.1007/s11269-013-0321-4

Trotter, L., Saft, M., Peel, M.C., Fowler, K.J.A., 2023. Symptoms of Performance Degradation During Multi-Annual Drought: A Large-Sample, Multi-Model Study. Water Resour. Res. 59. https://doi.org/10.1029/2021WR031845

UK Climate Resilience Programme, 2023. HIGH IMPACT SCENARIOS AND STORYLINES [WWW Document]. URL https://www.ukclimateresilience.org/projects/high-impact-scenarios-and-storylines/ (accessed 5.19.23).

UKCEH, 2023. Spatially explicit Projections of EnvironmEntal Drivers (SPEED) [WWW Document]. URL https://uk-scape.ceh.ac.uk/our-science/projects/SPEED (accessed 1.25.23).

Van Loon, A.F., 2015. Hydrological drought explained. WIREs Water 2, 359–392. https://doi.org/10.1002/wat2.1085

Van Loon, A.F., Laaha, G., 2015. Hydrological drought severity explained by climate and catchment characteristics. J. Hydrol. 526, 3–14. https://doi.org/10.1016/j.jhydrol.2014.10.059

Velázquez, J.A., Schmid, J., Ricard, S., Muerth, M.J., Gauvin St-Denis, B., Minville, M., Chaumont, D., Caya, D., Ludwig, R., Turcotte, R., 2013. An ensemble approach to assess hydrological models' contribution to uncertainties in the analysis of climate change impact on water resources. Hydrol. Earth Syst. Sci. 17, 565–578. https://doi.org/10.5194/hess-17-565-2013

Vetter, T., Reinhardt, J., Flörke, M., van Griensven, A., Hattermann, F., Huang, S., Koch, H., Pechlivanidis, I.G., Plötner, S., Seidou, O., Su, B., Vervoort, R.W., Krysanova, V., 2017. Evaluation of sources of uncertainty in projected hydrological changes under climate change in 12 large-scale river basins. Clim. Change 141, 419–433. https://doi.org/10.1007/s10584-016-1794-y

Wilby, R.L., Harris, I., 2006. A framework for assessing uncertainties in climate change impacts: Low-flow scenarios for the River Thames, UK. Water Resour. Res. 42, 1–10. https://doi.org/10.1029/2005WR004065

Yang, X., Magnusson, J., Xu, C.Y., 2019. Transferability of regionalization methods under changing climate. J. Hydrol. 568, 67–81. https://doi.org/10.1016/j.jhydrol.2018.10.030

D: Groundwater droughts in England: a review of current understanding and knowledge gaps, and opportunities and priorities for future research

John P. Bloomfield and Christopher R. Jackson, British Geological Survey.

Overview

The Environment Agency commissioned a series of essays in 2022 aiming to review the state of understanding of drought as it affects different components of the terrestrial water cycle in England, including lessons from the past and both current and future impacts, to provide new insights and a common basis for management and planning. The essays collate the latest understanding about drought in England, with reference to the wider literature. This essay is part of the series and addresses droughts in the groundwater component of the terrestrial water cycle in England. A series of knowledge gaps are identified that frame future research needs.

Groundwater droughts, which have had major impacts on groundwater resources in England, are episodic and primarily driven by deficits in precipitation. However, their characteristics are a function of a wide range of catchment characteristics, including hydrogeology, and consequently they are challenging to forecast. Future predictions of groundwater drought are complicated further by long-term changes in the environment and changes in water resource management.

The essay is composed of four main sections, the first of which provides an overview of the state-of-the-art with respect to groundwater drought research. Because hydrological drought, including groundwater drought, is typically defined as a “lack of water” compared to some reference, how it is defined is an important question. Groundwater drought is often defined using groundwater levels, but there is a need to consider the value of different definitions and how these could be better used to support drought management and regulation. The methods that have been used to characterise groundwater drought generation, propagation, and termination are discussed, and issues relating to the inconsistent use of approaches to characterise drought highlighted. Both the catchment and hydrogeological controls on groundwater drought, and the spatio-temporal impacts of groundwater drought, are not well understood. The need for research on these topics is highlighted.

The second main section discusses the observational evidence for groundwater drought in England. Information on groundwater drought is often omitted from historical drought analyses and summaries. Whilst recent research has described past groundwater drought events, there is a need to gather groundwater-specific information related to droughts into a single resource. Furthermore, there has been no systematic longitudinal overview and analysis of groundwater droughts in the observed record for England and how they related to the driving climatology. The studies that have been undertaken based on the observed record in England, and the related data and information services, are discussed.

The penultimate main section considers modelling and forecasting of groundwater droughts and their impacts. Compared with other types of hydrological drought, there is only a limited international peer-reviewed literature related to modelling of aspects of groundwater droughts. However, within England there is a body of work undertaken by water companies and the consultancy sector to better constrain groundwater availability during droughts. The modelling studies that have been published are diverse in both their research aims and the approaches and methods they adopted. So, it is difficult to identify consistent research themes related to groundwater modelling of droughts. However, there is a clear need for a thorough review of modelling requirements related to groundwater-aspects of drought in England with particular emphasis on the most appropriate approaches to integrate groundwater processes into linked models of drought dynamics, and the observational data needed to evaluate their skill. There is certainly a lack of knowledge about hydrogeological and river-aquifer interaction processes operating during drought because of the limited availability observational data at the required spatio-temporal scales. There is also a need to produced better integrated models that capture the processes operating in, interactions between, and responses of different components of the terrestrial water cycle, and which couple with representations of the anthropogenic use and management of water.

The final main section summarises and tabulates the knowledge or research gaps identified in the essay. For each knowledge or research gap, opportunities for future research are identified and given a relative priority.

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Summary

The Environment Agency commissioned a series of essays in 2022 aiming to review the state of understanding of drought as it affects different components of the terrestrial water cycle in England, including lessons from the past and both current and future impacts, to provide new insights and a common basis for management and planning. The essays collate the latest understanding about drought in England, with reference to the wider literature. This essay is part of the series and addresses droughts in the groundwater component of the terrestrial water cycle in England. A series of knowledge gaps are identified that frame future research needs.

Groundwater droughts, which have had major impacts on groundwater resources in England, are episodic and primarily driven by deficits in precipitation. However, their characteristics are a function of a wide range of catchment characteristics, including hydrogeology, and consequently they are challenging to forecast. Future predictions of groundwater drought are complicated further by long-term changes in the environment and changes in water resource management.

The essay is composed of four main sections, the first of which provides an overview of the state-of-the-art with respect to groundwater drought research. Because hydrological drought, including groundwater drought, is typically defined as a "lack of water" compared to some reference, how it is defined is an important question. Groundwater drought is often defined using groundwater levels, but there is a need to consider the value of different definitions and how these could be better used to support drought management and regulation. The methods that have been used to characterise groundwater drought generation, propagation, and termination are discussed, and issues relating to the inconsistent use of approaches to characterise drought highlighted. Both the catchment and hydrogeological controls on groundwater drought, and the spatio-temporal impacts of groundwater drought, are not well understood. The need for research on these topics is highlighted.

The second main section discusses the observational evidence for groundwater drought in England. Information on groundwater drought is often omitted from historical drought analyses and summaries. Whilst recent research has described past groundwater drought events, there is a need to gather groundwater-specific information related to droughts into a single resource. Furthermore, there has been no systematic longitudinal overview and analysis of groundwater droughts in the observed record for England and how they related to the driving climatology. The studies that have been undertaken based on the observed record in England, and the related data and information services, are discussed.

The penultimate main section considers modelling and forecasting of groundwater droughts and their impacts. Compared with other types of hydrological drought, there is only a limited international peer-reviewed literature related to modelling of aspects of groundwater droughts. However, within England there is a body of work undertaken by water companies and the consultancy sector to better constrain groundwater availability during droughts. The modelling studies that have been published are diverse in both their research aims and the approaches and methods they adopted. So, it is difficult to identify consistent research themes related to groundwater modelling of droughts. However, there is a clear need for a thorough review of modelling requirements related to groundwater-aspects of drought in England with particular emphasis on the most appropriate approaches to integrate groundwater processes into linked models of drought dynamics, and the observational data needed to evaluate their skill. There is certainly a lack of knowledge about hydrogeological and river-aquifer interaction processes operating during drought because of the limited availability observational data at the required spatio-temporal scales. There is also a need to produced better integrated models that capture the processes operating in, interactions between, and responses of different components of the terrestrial water cycle, and which couple with representations of the anthropogenic use and management of water.

The final main section summarises and tabulates the knowledge or research gaps identified in the essay. For each knowledge or research gap, opportunities for future research are identified and given a relative priority.

1. Introduction

The Environment Agency (EA) commissioned a series of essays in November 2022 with the aim of reviewing the state of understanding of drought as it affects different components of the terrestrial water cycle in England, including lessons from the past and both current and future impacts, to provide new insights and a common basis for management and planning. The essays consist of collations of the latest information and understanding about drought in England, with reference, where relevant, to the wider UK and international literature. The essays seek to answer the following high-level questions:

- What is known about droughts?
- What is unknow about droughts but may be pertinent to current water resource management approaches and to future planning and decision making in England?
- How can understanding of drought be improved to address current and future needs?

Drought is a phenomenon that affect the entire terrestrial water cycle and can have wide-reaching environmental and societal impacts and feedbacks and consequently are best understood in a whole-system context. Notwithstanding that, this essay focusses on droughts in the groundwater component of the terrestrial water cycle in England.

Groundwater droughts have had major impacts on groundwater resources in England and on the livelihoods of those who use groundwater resources. They can also have significant impacts on the sustainability and health of groundwater-dependent terrestrial ecosystems, including, for example, high baseflow rivers and groundwater-fed wetlands. Groundwater droughts are episodic and primarily driven by deficits in precipitation, although they may be exacerbated by heatwaves (Bloomfield et al., 2019). However, their onset, development and termination are a function of a wide range of catchment characteristics, including the nature of the underlying hydrogeological systems, and consequently they are challenging to forecast and predict. Future predictions of groundwater drought are complicated further by long-term changes in the environment. Changes in water resource management practices may also influence the dynamics of groundwater droughts. Changes in climate are already affecting the frequency, magnitude, intensity and duration of episodes of groundwater droughts (Bloomfield et al, 2019) and climate change is expected to continue to change the nature and impacts of groundwater droughts in an uncertain manner into the future (Jackson et al, 2015; Watts et al., 2015). This essay will review the literature on all these aspects of groundwater drought.

The aims of this essay on groundwater drought are three-fold, to: document the current state-of-the-art understanding of groundwater droughts in England based on the peer-reviewed literature; highlight key gaps in knowledge of groundwater drought processes, data, and forecasting and modelling, with a particular emphasis on those gaps that have implications for current approaches to groundwater drought management and to future planning and decision making in England related to groundwater droughts; and, to identify opportunities and priorities for future research.

The essay is structured as follows: Section two is an overview of the state-of-the-art with respect to groundwater drought research with a focus on temperate groundwater systems in England. It includes discussion of definitions and characterisation of groundwater droughts, drought propagation and termination processes, and a brief note on the impacts of groundwater drought. Section three is an overview of observational evidence for groundwater droughts in England. This includes a summary of the data and evidence available to characterise groundwater droughts, and an overview of major episodes of groundwater drought in the observational and reconstructed records from the present back to ~1900, including documented changes associated with climate change and other anthropogenic changes to catchments. The section is concluded with a note on currently available data and information services related to groundwater droughts in the UK. Section four is an overview of groundwater modelling and forecasting methodologies in the literature related to groundwater droughts and an assessment of the current challenges and limitations related to the modelling of groundwater droughts. The section also includes a note on approaches to modelling changes in groundwater droughts under climate change and other environmental impact modelling studies. Section five summarises the knowledge gaps identified in the previous sections and describes the opportunities and priorities for future groundwater drought research. It is concluded with a series of recommendations.

2. Groundwater drought

2.1. Definitions of groundwater drought

Drought, including groundwater drought, is an episodic natural hazard that impacts people globally, both directly and indirectly, for example though the effects on water supplies, food production, industry, power production, and recreation (Mishra and Singh, 2010). The environmental impacts of drought are equally profound and diverse, including changes in ecosystem structure and function over a wide range of spatio-temporal scales (Van Loon, 2015).

Droughts have been conceptualised as effecting different components of the water cycle, e.g. meteorological drought and hydrological drought, the latter including groundwater drought (Mishra and Singh, 2010), or by their impact on different compartments of the environment or society, e.g. ecological drought (Slette et al., 2019) or agricultural drought Orimoloye (2022).

Groundwater drought is usually conceptualised as a type of hydrological drought (Van Loon 2015), where, citing Tallaksen and Van Lanen (2004), Van loon (2015) defined hydrological drought as: “a lack of water in the hydrological system, manifesting itself in abnormally low streamflow in rivers and abnormally low levels in lakes, reservoirs, and groundwater”.

An important element of this definition is that hydrological drought, including groundwater drought, is a relative, not absolute concept, i.e. groundwater drought is defined as a “lack of water” compared to some reference state, or as an “abnormally low level” of groundwater: again compared to some measure of normal (Chang and Teoh, 1995; Eltahir and Yeh, 1999).

Note that some workers have suggested that droughts may also be a linked hydro-meteorological-social construct (Lange et al., 2016) or that they should be framed by large-scale, long-term environmental changes such as those associated with the Anthropocene (Van Loon et al, 2016). Both of these concepts are out of scope for this essay and the focus is on meteorologically driven groundwater droughts in the observational record and as reconstructed and forecast out to centennial time scales.

As groundwater systems are affected by droughts, initially groundwater recharge decreases leading to supressed groundwater levels and finally to reduced groundwater discharge. Consequently, some researchers have defined groundwater drought in terms groundwater recharge droughts, groundwater storage droughts, and groundwater discharge droughts, the latter including the concept of groundwater baseflow droughts (Hellwig and Stahl, 2018; Hellwig et al, 2021). Peters and co-workers developed this concept in a series of landmark papers in the early 2000s when simulating groundwater drought distribution and propagation in an idealised Chalk catchment (Peters et al., 2003; 2005) and applied this conceptualisation to the Pang catchment, Berkshire (Peters et al., 2006). These definitions are occasionally used, for example:

- Barthel et al. (2021) identified the importance of understanding recharge in the context of the typically small, low storage groundwater systems in Sweden and highlighted the challenge that groundwater recharge studies are typically site specific and that transferability of findings from one setting to another is challenging.
- Wossenyeleh et al. (2022) investigated groundwater recharge droughts in the context of wider drought propagation process in semi-arid regions of Ethiopia.
- The concept of a groundwater discharge drought has been particularly useful to those working on karst systems with major spring outflows. For example, Fiorillo and Guadagno (2010, 2012) defined groundwater drought in a karst system in southern Italy based on low flows in spring discharge timeseries.

However, because the negative impacts of groundwater storage depletion can be felt well before total groundwater storage is depleted (Van Lanen and Peters, 2000), most studies of groundwater drought have focussed on droughts as defined by abnormally low groundwater levels in unconfined aquifers (Mishra and Singh, 2010). An additional benefit of characterising groundwater drought based on changes in groundwater level is that these data are typically readily available for most groundwater systems. For simplicity, throughout the rest of this essay 'groundwater drought' should be taken to refer to droughts defined by groundwater (heads) levels in unconfined aquifers unless otherwise stated.

Knowledge gap 1: Which, if any, EA regulatory functions or processes require or would benefit from a formal definition of groundwater drought? Is there a need or benefit in discriminating between groundwater recharge, storage and discharge droughts in different regulatory settings? How would such formal definitions of groundwater drought relate to wider formal definitions of drought that may also be used in a regulatory context?

2.2. Overview of methods and approaches used to characterise and quantify groundwater droughts

There are two broad approaches to quantifying hydrological droughts: threshold level approaches and standardised index approaches. In addition, there are other simpler approaches, such as simple ranking or percentile approaches (Chang and Teoh, 1995), that are typically used in reporting groundwater resource status including during periods of groundwater drought (e.g. as part of BGS-UKCEH's Monthly Hydrological Summaries (UKCEH, 2023a), or the EA's Water Situation Reports (Environment Agency, 2022). However, as Van Loon (2015) notes, there is no 'best' hydrological drought index or approach and the choice of approach or use of specific index and the detail of their implementation are important as they can affect the results and conclusions of a given study or analysis.

The threshold level approach enables droughts and their characteristics to be defined for hydrological time series for a give threshold level which may either be fixed or time variable (Beyene et al., 2014). As the threshold level approach uses absolute levels it is possible to quantify absolute drought magnitudes that can directly inform drought management decisions. It also has the benefit that thresholds can be used as ‘trigger levels’ as part of drought management plans and, consequently, Van Loon (2015) recommends that “ideally … threshold level should be related to drought impacted sectors/systems, e.g. irrigation water requirements”. For example, absolute levels of groundwater under drought conditions is an important factor in the healthy functioning of groundwater dependent terrestrial ecosystems (GWDTEs) (Wossenyeleh et al., 2021) and in this context threshold approaches to quantifying the impact of groundwater droughts on wetlands might be considered to have more utility than standardisation approaches for a given site. However, information related to water resource management to constrain suitable thresholds (for example, in the case of a GWDTE a critical groundwater level required to sustain good ecosystem status) is commonly lacking and it is much more common for studies to set thresholds using percentiles (Heudorfer and Stahl, 2017).

The threshold level approach using percentile thresholds has been used in studies of hydrological drought propagation, including propagation of drought through groundwater systems (Peters et al., 2006; Tallaksen et al., 2009). However, there are disadvantages to this approach. Unlike the standardised index approach described below, as no standard drought classes are calculated it is problematic to directly compare drought response between sites or over the large areas effected by major episodes of drought with large spatio-temporal footprints and particularly across highly heterogeneous catchments. In addition, Van Loon (2015) notes “subjective choices cannot be avoided, for example on the threshold level [percentile] to use”. In a comparison of different threshold level methods (different threshold percentiles and constant threshold versus temporally variable thresholds) for drought propagation analysis in Germany, Heudorfer and Stahl (2017) concluded that there is potential for diverging inferences in drought phenomena depending on the details of the chosen threshold method. For example, using the variable threshold method they found a substantial increase in short droughts and a minor increase in long droughts relative to a constant threshold.

A few groundwater studies have used the threshold approach for groundwater drought definition and characterisation. For example, Chang and Teoh (1995) assessed the sensitivity of modelled drought characteristics based on a range of constant percentile thresholds. Peters and co-workers used the approach to examine spatial variation in the propagation of modelled droughts in the Chalk of the Pang catchment, UK (Peters et al., 2003; 2006), and the performance of models in predicting drought propagation to groundwater (Peters et al. 2005). Fendekova and Fendek (2012) applied the threshold approach to groundwater baseflow in the Nitra River, Slovakia to identify droughts. Wossenyeleh et al. (2021) recently applied the variable threshold approach to a study drought propagation (groundwater recharge and groundwater discharge) and its impact on groundwater hydrology of a small (~10km2) wetland (Doode Bemde, Belgium). However, due to the previously mentioned challenges associated with applying the approach, use of the threshold level method to characterise groundwater droughts has been relatively limited and the use of standardised index approaches are much more common.

Standardised drought indices consist of measures of departures or anomalies from the ‘normal’ situation. Such indices have the benefit of enabling regional comparison of droughts, important since major droughts have large spatio-temporal footprints, and the comparison of drought status between different components of the terrestrial water cycle where standardised drought indices are estimated for precipitation, groundwater level and streamflow. However, unlike the threshold approach to defining episodes of drought, the standardised approach has the drawback that the severity of a drought event is expressed only in relative terms and the approach has reduced utility where operational decisions may require absolute values of water deficits compared with ‘normal’ conditions (i.e., deficit volumes) to be identified. In addition, where vertical heterogeneity in aquifer characteristics effects groundwater heads, or in confined aquifers where there is a spatial- and or temporal-disconnect with the driving climatology standardised groundwater indices may be of limited use in understanding the response of groundwater systems to meteorological droughts. As with the threshold method, there are other additional methodological challenges summarised below.

Standardised drought indices follow from the work of McKee et al. (1993) who developed the Standardized Precipitation Index (SPI) (World Meteorological Organization, 2012). The SPI uses long-term records and fits distributions to monthly values of precipitation. These fitted distributions for each month are then re-scaled or transformed to a normal distribution with a mean of zero and a unit standard deviation and the standardised monthly values recombined to give the full standardised time series. SPI is typically computed over a range of time scales (accumulation periods), e.g. 1, 3, 6, 12 and 24 months, to enable the effects of accumulating precipitation deficits (or excesses) to be investigated (Van Loon, 2015). There are two main methodological challenges related to the SPI. The approach requires long records, typically of 30 years or more of monthly precipitation totals, and the choice of fitted probability distributions can have a significant effect on the resulting estimates of SPI (Stagge et al., 2015a; Van Loon, 2015). Notwithstanding these challenges, it is one of the most widely used drought indices and has inspired related indices such as the Standardized Precipitation and Evapotranspiration Index (SPEI) (Vincente-Serrano, et al., 2009), the Standardised Streamflow Index (SSI) (Vicente-Serrano et al., 2012), and the Standardised Groundwater level Index (SGI) (Bloomfield and Marchant, 2013).

There are a number of standardised groundwater indices that have been used in groundwater drought and water resources studies. For example, Bhuiyan et al, (2006) described a Standardised Water-level Index (SWI) developed to quantify seasonal water stress in Rajasthan, India. The SWI is estimated by dividing the difference between the seasonal water level and its long-term seasonal mean by the standard deviation of the observations. Seasonal water levels, rather than monthly levels, are used since cycles in regional groundwater resources in Rajasthan are dominated by the relative strength of the seasonal monsoon. Mendicino et al. (2008) developed the Groundwater Resource Index (GRI) to normalise the retention of groundwater in a water balance model of Calabria, Italy, and estimated the GRI as the difference between the modelled monthly groundwater retention and the long-term mean value divided by the standard deviation of the modelled values. Fiorillo and Guadagno (2010; 2012) applied the SPI method to karstic spring discharges from sites in Southern Italy to identify and characterise groundwater droughts. More recently, Thomas et al. (2017) have extended the water balance approach of

Mendicino et al. (2008) to apply it to evaluate groundwater droughts based on data from NASA's Gravity Recovery and Climate Experiment (GRACE) satellite mission, where normalized GRACE-derived groundwater storage deviations are defined as the GRACE Groundwater Drought Index (GGDI).

However, the most extensively used groundwater index in drought studies is the Standardised Groundwater level Index (SGI) (Bloomfield and Marchant, 2013). The SGI is similar to the SPI although SGI uses a non-parametric approach to rescaling monthly groundwater levels. A non-parametric approach to standardisation is preferable for groundwater level time series since Bloomfield and Marchant (2013) demonstrated that typically there was no consistent distributions that could be fitted to groundwater level data between sites or even between months at a given site. Note that notwithstanding this observation, some studies still fit distributions to estimate SGI in a manner similar to the SPI method (e.g. Guo et al., 2021 fitted a log normal distribution to standardise monthly groundwater levels in the USA). Marchant and Bloomfield (2022) have further refined the SGI method by combining it with groundwater level data pre-processing and modelling steps that remove data outliers and generate a regular monthly groundwater level time series prior to estimation of the SGI. They illustrated their workflow with data from six sites in the UK (Marchant et al., 2022). In addition, methods have been developed to lengthen (reconstruct) groundwater level time series to provide sufficiently long time series to apply the SGI method (Jackson et al., 2016) (see section 4).

In the absence of systematic observations of groundwater levels and SGI data, the use of SPI to infer groundwater drought status has been investigated using data from Germany and the Netherlands by Kumar et al. (2016). However, due to catchment and aquifer specific response of groundwater systems to precipitation deficits, it was not possible for Kumar et al. (2016) to find a uniform SPI accumulation period (n) that enabled a satisfactory correlation to be established between SPIn and SGI and they concluded that there is a "need for more groundwater observations and accounting for regional hydrogeological characteristics in groundwater drought monitoring".

As noted previously, although threshold and standardisation methods are usually used to quantify groundwater droughts, other simpler approaches, such as simple ranking or percentile approaches are used to report groundwater resource status including during times of drought (UKCEH, 2023a; Environment Agency, 2022). These approaches lend themselves to simple graphical presentations of resource status and are typically developed for non-specialist users to aid the communication of hydrogeological status, including droughts, but generally have limited applicability to the quantification of groundwater droughts.

Regardless of whether groundwater droughts are identified using a threshold or standardised index approach, individual episodes of groundwater drought can be extracted from groundwater level or standardised time series and quantified by their start and end, duration, magnitude (total drought deficit in absolute or standardised terms), and intensity (Mishra and Singh, 2010; Van Loon, 2015; Parry et al. 2016). Characteristics of multiple groundwater droughts experienced at an observation well can be investigated as can the relative frequency of droughts (e.g. Bloomfield et al., 2015, Figure 11), and changes in drought characteristics over time can be investigated (e.g. Bloomfield et al. 2019). These groundwater drought characteristics have been used to explore and quantify major episodes of groundwater drought at a site or across multiple sites to investigate responses of different catchments and aquifers to regional driving climatology (Bloomfield and Marchant, 2015; Marchant and Bloomfield, 2018) as described in the following sections.

Knowledge gap 2: No guidance is currently available for EA staff and their contractors regarding the different methods to characterise and quantify groundwater droughts, including the pros and cons of the different methods, the implications for the nature of any groundwater droughts so defined, and the suitability of a given method for specific EA regulatory procedures or requirements.

Knowledge gap 3: Within water company drought management plans in England, there is no consistent approach to groundwater drought quantification and characterisation, and in particular to the setting of drought trigger levels. Consequently, it is not possible to assess if water companies are using consistent trigger levels in response to an episode of drought. What are the implications of this for drought management planning; what would be the benefits, if any, of encouraging the use of consistent definitions of groundwater droughts across water company drought management plans; and, what would be the most suitable approaches to drought definition, e.g. use of SGI?

2.3. Groundwater drought generation, propagation and termination

Hydrological droughts are primarily generated as a result of deficits in precipitation (Mishra and Singh, 2010) but may be modulated by other factors such as periods of abnormally high evapotranspiration. The generation of groundwater droughts is a function of the propagation of precipitation deficits to groundwater recharge deficits and so is a function of catchment setting, for example the depth to groundwater level within a catchment (Peters et al., 2006), soil properties and antecedent conditions such as soil moisture conditions (Van Loon, 2015), and of unsaturated and saturated aquifer characteristics, i.e. the responsiveness of aquifers to those propagating deficits (Bloomfield et al, 2015). For example, for relatively high storage aquifers that are dominated by autumn and winter recharge groundwater droughts are typically generated by two or more relatively dry annual recharge seasons and usually become evident in late winter or spring of the second year. In responsive, typically low storage aquifers groundwater recharge droughts may be generated within a season in response to relative short-term precipitation deficits and may occur at any time of the year. The ‘memory’ within a given groundwater system, as expressed by the autocorrelation structure of the groundwater level or SGI time series, is a good indicator of the relative nature of groundwater drought generation. Bloomfield and Marchant (2013) demonstrated that fewer droughts are generated in aquifers with relatively long SGI autocorrelation ranges compared to those aquifers with shorter SGI autocorrelation ranges.

Changnon (1987) and Eltahir and Yeh (1999) introduced the concept of drought propagation through components of the terrestrial water cycle, including through groundwater, and Van Loon (2015, Figure 3) provided a detailed overview of the concept. The conceptual model describes the propagation of a water deficit from rainfall deficit (meteorological drought) through soil moisture deficit (soil moisture drought) to deficits in groundwater recharge, to low groundwater storage, and to low groundwater discharge (groundwater drought). Streamflow droughts may be associated with the early stage of the meteorological and soil moisture droughts in low baseflow systems where run-off is dominated by surface and shallow soil processes but may be associated with later stages of droughts in high baseflow systems where the majority of the streamflow is provided by groundwater discharge.

As Eltahir and Yeh (1999) described, while discharge from unconfined aquifers to streams as baseflow provides a mechanism for the dissipation of wet anomalies, the nonlinear dependence of groundwater discharge on groundwater levels may explain why droughts can be significantly more persistent than floods. They attributed the nonlinearity to increasing disconnection between unconfined aquifers and stream networks as droughts develop. However, Bloomfield et al. (2018a) noted the nonlinear dependence of groundwater discharge on groundwater levels is also present in catchments when there is effectively no groundwater surface water interaction and that the nonlinear behaviour can also be explained by factors such as vertical variations in aquifer properties.

As drought deficits propagate, the groundwater system acts as a low pass filter changing the nature of the original drought signal and, relative to the driving meteorological drought, groundwater droughts, particularly those defined by groundwater levels and spring discharges, may be pooled, lengthened, lagged and attenuated (Van Loon, 2015, Figure 4). This phenomenon can be seen for example in the plot of SPI, SSI (Standardised Streamflow Index) and SGI for the regional English Lowlands series (Folland et al., 2015, Figure 5). However, to date there have been no integrated observational studies of the spatio-temporal dynamics of drought propagation in permeable catchments representative of much of England. In this context, it is worth noting that during the NERC-funded Historic Droughts project (UKCEH, 2023b), the project team were unable to identify a single gauged catchment in the UK with suitable long, co-located observational records of driving meteorology, soil moisture, groundwater, and surface flows to investigate and characterise drought propagation through the terrestrial water cycle.

The termination of hydrological droughts, including groundwater droughts, has received relatively little attention compared with drought generation and propagation. Following a review of the drought termination literature, Parry et al. (2016) provided the first systematic framework to define and characterise drought termination. They framed the concept as a period during which a drought ends (rather than a specific date) and defined drought termination start and end dates, durations, rates and magnitudes (Parry et al., 2016, Figure 3). They applied their methodology to the drought of 2010-2012 in the UK and noted the relatively rapid termination of that drought in the Chalk (four to six months) compared with a more prolonged termination in Permo-Triassic sandstones. Parry et al. (2018) subsequently used historical data comprising ‘current rate’ and ‘historical ensemble’ approaches to assess the prospects of groundwater level recovery and tested this using data from England and Wales for 2017-2018. Parry et al. (2018) considered neither approach satisfactory on their own. For example, if current events are outside the envelope of historical events than the historical ensemble approach cannot provide any information on groundwater drought termination and recovery, and if a groundwater drought has yet to enter the drought termination phase then the current rate approach cannot provide any useful information on drought termination. Lumped groundwater models driven by forecasts of precipitation and temperature offer a better approach to the short-term (up to a season ahead) prediction of groundwater drought termination (Mackay et al., 2015) (see section 4). In an analysis of groundwater recovery from drought in unconfined aquifers across the USA, Schreiner-McGraw and Ajami (2021) documented average recovery times of three years (and up to 15 years in some aquifers) and found that recovery times were primarily a function of drought intensity at the beginning of the precipitation drought and the mean annual recharge.

Knowledge gap 4: There is no high storage catchment in England with long, co-located observational records of driving meteorology, soil moisture, groundwater recharge, storage and discharge, and surface flows to investigate processes associated with drought initiation, propagation through the terrestrial water cycle, and termination in a groundwater-dominated catchment.

2.4. Catchment and aquifer controls on groundwater droughts

Meteorological droughts are dependent on large-scale atmospheric circulation patterns that typically cover large areas, however, hydrological droughts, including groundwater droughts, usually affect smaller areas as they are influenced by catchment characteristics (Van Loon, 2015). For example, Zaidman et al. (2002) and Hannaford et al. (2011) documented this effect for streamflow droughts in NW Europe. In the context of groundwater droughts, spatial variability in aquifer characteristics may also result in reduced spatial footprints and spatio-temporally varying groundwater drought characteristics.

The National Rivers Authority (a precursor agency to the Environment Agency) commissioned the BGS to assess how much water was stored in the Chalk aquifer (NRA, 1993). The BGS developed a national 3D model of the Chalk to estimate the total volume of water in the Chalk aquifer, the volume stored within the zone of natural water level fluctuation, and the volume between minimum water level but above Ordnance Datum. This model was built using and shown to be sensitive to spatially and depth varying values of specific yield and specific storage for the Chalk. Although it was not used to estimate available groundwater resource under drought, an updated model based on a similar modelling approach would potentially be a useful management and communication tool during future droughts.

Bloomfield and Marchant (2013) demonstrated a positive approximately linear relationship between maximum drought duration and the autocorrelation range (or ‘memory’) of the

SGI for a given site. Although they only analysed 14 sites from the Chalk, Permo-Triassic Sandstone, Lower Greensand and Lincolnshire Limestone aquifers, they noted that Permo-Triassic Sandstone sites typically had relatively long SGI autocorrelations and experienced long groundwater droughts, whereas there was a relatively wide range of SGI autocorrelations and characteristic drought lengths for the Chalk sites analysed. For Chalk sites, Bloomfield and Marchant (2013) also demonstrated a positive linear relationship between the autocorrelation range or groundwater 'memory' and unsaturated zone thickness. Note that since unsaturated zone thickness is also a function of the position of an observation borehole in the catchment, with thicker unsaturated zone present in the headwaters of Chalk catchments, this is consistent with potential variations of in-catchment Chalk drought characteristics modelled by Peters et al. (2006). Peters et al. (2006) noted that models of the spatial distribution of groundwater droughts in hydraulic heads predict that short groundwater droughts are more likely near streams and discharge points, whereas longer droughts are more likely near groundwater divides.

In a study of 74 standardised hydrographs from the unconfined and confined Chalk, Lincolnshire Limestone, and the confined Spilsby Sandstone aquifers of Lincolnshire, Bloomfield et al. (2015) demonstrated spatially coherent responses between the different unconfined aquifers in response to a series of common forcing meteorological droughts. For example, the sites in the unconfined Chalk aquifer exhibited notably fewer, longer, less intense groundwater droughts than sites in the Lincolnshire Limestone.

In an analysis of SGI time series for 948 sites on the Chalk of England, Marchant and Bloomfield (2018) identified seven clusters showing similar temporal responses features. Two of these clusters were interpreted as being associated with long-term trends associated with local over abstraction and groundwater rebound, and a third showed no significant temporal trend or correlation with driving climatology. The remaining four clusters were spatially coherent and were interpreted as primarily reflecting regional differences in the hydrogeology of the Chalk aquifer that consequently resulted in regional differences in response to meteorological droughts. They demonstrated, for example, the relatively long autocorrelation of the SGI time series for the drift covered Chalk of East Anglia and Lincolnshire and hence the tendency to fewer but longer droughts in the region compared with more southern Chalk. However, the principal novelty of the work was that for the first time it was possible to map on a monthly timestep the status of groundwater in the Chalk aquifer at a national scale. For example, Marchant and Bloomfield (2018) mapped the spatial variation in groundwater drought defined by SGI from August 1975 to February 1977 (Marchant and Bloomfield, 2018, Figure 14). This mapping showed that during the 1975-77 drought, groundwater drought was most severe in the Chalk of the Wessex Basin and Berkshire Downs, in part due to the relatively rapid response of this part of the Chalk (compared to, for example, the Chalk of East Anglia) to the driving meteorological drought.

Knowledge gap 5: It is evident from several studies that groundwater response to large meteorological droughts is spatially variable, particularly in the Chalk aquifer, and that there is spatial variation in the sensitivity of groundwater to meteorological droughts of different durations and magnitudes. However, unlike groundwater flooding, there is no mapping of the spatial variation in groundwater susceptibility or sensitivity to drought across England. There is currently no national volumetric assessment of Chalk storage and the sensitivity of storage to drought.

2.5. Impacts of groundwater drought

The impacts of groundwater droughts are varied and can be significant. They include, for example, reduced yields from public and private supply boreholes (Ascott et al., 2019), reduced baseflow to groundwater supported streams and wetlands with adverse impacts on ecology and reduced amenity value, reduced availability of groundwater for irrigation, and reduced availability for industrial supply (Ascott et al., 2021). However, there is very limited systematic information to quantify the impacts of groundwater droughts (as opposed to meteorological or other hydrological droughts).

At the European scale there is some systematic information on the impacts of meteorological droughts in the European Drought Impacts Inventory (EDII) (Stahl et al., 2015; European Drought Centre, 2022) in 15 pre-defined categories, including, for example: public water supply, agriculture and livestock farming, freshwater ecosystems, and tourism. In the UK, a series of data outputs from the Historic Droughts project including inventories of water resources, regulations, agricultural media and newspaper articles from the 1800s to 2014 are available through the UKCEH website (https://www.ceh.ac.uk/our-science/projects/historic-droughts-data) and the UK Data Service (https://ukdataservice.ac.uk/). These contain some references to the impacts of groundwater droughts but that information has not been systematically extracted and analysed.

Stagge et al. (2015b) used logistic regression to model relationships between SPI (and SPEI) and drought impacts documented in the EDII for data from five European states including the UK. They found that over all agricultural impacts were best explained by 2 to 12 SPI anomalies, whereas public water supply and freshwater ecosystem impacts were explained by more complex combinations of short (1–3 month) and seasonal (6–12 month) SPI anomalies. They also noted that “among the five European countries analysed, the United Kingdom consistently had the slowest drought response, likely due to its slow responding groundwater storage [of the Chalk]”.

In the UK work on the impacts of groundwater drought have to date typically focussed on the reduced availability of groundwater for public supplies. Beeson et al. (1997) and Misstear and Beeson (2002) developed a methodology to enable the assessment of declines in borehole yield under drought. The methodology requires historical data on drawdown at different pumping rates, for example from step-drawdown tests or operational monitoring, preferably during groundwater droughts. These yield-drawdown data are used to construct a drought-bounding curve that indicates what the reliable pumping rate can be expected to be for any given drawdown during droughts. If this yield-drawdown drought-bounding curve intersects a critical groundwater level constraint (e.g. such as a pump intake level or top of a major inflow horizon), then the intersection point defines the ‘deployable output’ for that abstraction borehole. Ascott et al. (2019) have recently used this methodology to explore the relative importance of aquifer heterogeneity and climate change on theoretical drought yields for representative boreholes and found that variation in hydraulic conductivity is as significant a control on drought yields as future UK climate scenarios. One of the practical challenges, however, with application of the yield-drawdown curve approach is that it is dependent on yield-drawdown data being available from step tests performed under suitable (extreme) drought conditions. For many abstraction boreholes these data are very limited and typically only available for moderate drought episodes.

Knowledge gap 6: There is no typology for groundwater drought impacts for the UK (or internationally) and no institution or group is actively collating and maintaining a record of groundwater drought impacts in an inventory either for regulatory or research purpose. This is an essential precursor to any future quantitative assessment of the links between driving climatology, groundwater drought status and impacts of groundwater drought.

Knowledge gap 7: There is generally a poor understanding of borehole yields during episodes of major droughts.

3. Observational evidence for groundwater droughts in England

Information related to historical episodes of groundwater drought in England is typically available through reports and peer-reviewed papers describing individual major meteorological and hydrological droughts (e.g. Marsh and Parry, 2012), although information on groundwater aspects of droughts is often omitted altogether (see for example the description of the 2012 drought by Kendon et al., 2013). To date groundwater-specific information related to droughts has not been gathered together into a single resource with accompanying groundwater-specific narrative. However, the drought inventory produced by the recent NERC-funded Historic Droughts project does include description of the groundwater aspects of 17 major episodes of drought in the UK since 1890 (UKCEH, 2023b). As part of that project BGS produced reconstructed monthly groundwater levels and SGI time series at 54 sites across the UK for the period 1890 to 2015 (Bloomfield et al., 201b; 2018c), however, episodes of drought were not extracted from those reconstructed time series and analysed.

The only studies to date to extract and analyse groundwater droughts in England have been by BGS (Bloomfield and Marchant, 2013; Bloomfield et al, 2015; Marchant and Bloomfield, 2018). These have focussed on:

- the development of a methodology for groundwater drought definition (Bloomfield and Marchant, 2013);
- an investigation of aquifer and catchment and aquifer controls on groundwater drought propagation (Bloomfield et al, 2015; Marchant and Bloomfield 2018); and

- analysis of droughts at Chilgrove House and Dalton Holme and their association with climate warming (Bloomfield et al., 2019)

It is only the Bloomfield et al. (2019) study (described below) that attempted a systematic analysis of changes in groundwater drought characteristics with time, and then only at two observation boreholes on the Chalk.

Notwithstanding these comments, based on the Historic Droughts Inventory and information in the published literature it is evident that at least parts of England experience groundwater droughts at a frequency of typically once a decade, with groundwater droughts documented for the following periods: 1890-1910; 1913-1914; 1920-21; 1933-34; 1940s; 1962-64; 1973; 1975-76; 1984; 1988-93; 1995-98; 2003; 2004-06; and 2010-2012, and more recently, 2017-19; and 2022. Each of these episodes of groundwater drought is notable for its unique spatio-temporal character. For example, some having a northern or southern focus and some being relatively short-lived (less than a year) and others enduring for multiple years.

Knowledge gap 8: Despite the available data, there has been no systematic longitudinal overview and analysis of groundwater droughts in the observed record for England and how they related to the driving climatology. This is a significant omission given the importance of understanding and developing ‘benchmark groundwater droughts’ for reference for planning responses to future drought.

There have been few long-term, large-scale systematic observational studies on the effects of climate change or other anthropogenic changes to catchments on changes in groundwater droughts in England (Watts et al, 2015). This is due to a number of reasons, as follows:

- There is a paucity of long observational records to constrain temporal trends in hydrological time series, including groundwater levels (Jackson et al, 2015).
- By their very nature, groundwater droughts, like other droughts, are episodic and so there are typically few episodes in most groundwater level time series available to quantify and characterise changes in the nature of the droughts.
- Even if changes in groundwater drought characteristics can be identified there is the challenge of understanding the controls on any observed changes in groundwater drought characteristics. Long-term changes in groundwater drought characteristics, like any changes in groundwater dynamics, are potentially influenced by multiple factors, such as changes in:
    - land cover or land management practices,
    - in water resource management practices, or
    - changes in the driving climatology.

Notwithstanding these caveats, a couple of studies focussing on decadal and longer changes in groundwater droughts.

Changes in driving climatology may be associated with climate change or may simply reflect decadal or longer changes in large scale atmospheric circulation or teleconnections

(Rust, 2018; 2019; 2021; 2022). For example, Rust (2022) showed that there was a dominant 7.5 year periodicity in water resource extremes (including in groundwater levels) in the UK associated with the North Atlantic Oscillation (NAO), that the relationship between NAO and precipitation was non-linear over the analysis period of that study (1930 to 2020), and that multiannual NAO-precipitation relationships have modulated historical water resource anomalies "to an extent that is comparable to the projected effects of a worst-case climate change scenario [for the UK]".

Jackson et al. (2015) reported on Mann-Kendall trend tests for seven long SGI time-series from the Chalk aquifer and noted that there was some evidence to indicate long-term declines in groundwater levels in the Chalk of the UK but that as with other hydrological time-series (Wilby, 2006), it was not possible to ascribe the declines groundwater levels to in climate change. Subsequently, Bloomfield et al. (2019) analysed changes in groundwater drought characteristics at the two longest observation boreholes in the Chalk of the UK, at Chilgrove House and Dalton Holme and identified increases in the frequency and intensity of individual groundwater drought months, and increases in the frequency, magnitude, and intensity of episodes of groundwater drought. In addition, they documented an increasing tendency for both longer episodes of groundwater drought and for an increase in droughts of less than one year in duration. In the absence of long-term changes in land use/cover or in precipitation deficits over the same period, they inferred that the changing nature of groundwater droughts at these two sites is due to changes in evapotranspiration (ET) associated with anthropogenic warming and that this is due to the relatively thick capillary fringe associated with the Chalk.

Wendt et al. (2020) investigated the effect of groundwater use (abstraction) on groundwater droughts in the Chalk and Permo-Triassic sandstone aquifers using groundwater level data from the Chilterns, Lincolnshire, the East Midlands, and Shropshire. They documented two responses to abstractions: in systems where the long-term annual average groundwater abstractions are smaller than recharge an increase in the frequency of shorter groundwater drought events was observed, whereas where groundwater abstractions exceeded recharge a lengthening and intensification of groundwater droughts was observed. Consequently, they concluded that human-modified droughts differ in frequency, duration, and magnitude, depending on the long-term balance between groundwater use and recharge. The analysis period for the study was 30 years (1984-2014) and Wendt et al. (2020) noted that during that period regulated groundwater abstractions had reduced and that at the majority of sites analysed rising groundwater trends were observed.

Knowledge gap 9: There is evidence that a number of decadal-scale changes influence the occurrence and nature of droughts in England. These include decadal-scale changes in atmospheric circulation, anthropogenic climate change, and changes in water resource management practices in England. However, to date there has been no work to synthesise this evidence, to investigate the relative magnitude of these effects, to investigate if these effects are compounding (for example are they additive or multiplicative?), and if and how understanding of the change-drivers should inform future modelling, forecasting, planning and operational management activities related to groundwater droughts.

3.1. Data and information services

Drought monitors are available in a number of countries to inform decision making and planning responses to droughts, for example see the US Drought Monitor (https://droughtmonitor.unl.edu/), the European Drought Observatory (https://edo.jrc.ec.europa.eu/edov2/php/index.php?id=1000), and the Australian Drought Monitor (https://www.nacp.org.au/drought_monitor). Notably, all of these use complex, compound indices of 'droughtiness' that typically do not specifically include measures of groundwater drought status.

There is no dedicated equivalent drought monitor in the UK. However, UKCEH maintain the UK Water Resources Portal that provides near real-time access to water resource status information as raw data, including a limited number of groundwater observation boreholes, but importantly also includes the data rescaled to drought indices. The latter consists of area averaged SPI for accumulation periods of 1 to 24 months, SSI and SGI. Currently, SGI data are available for ~40 observation wells across the UK. Groundwater status information is also available in the Monthly Hydrological Summaries produced by UKCEH and BGS and the monthly Water Situation Reports for England produced by the Environment Agency, both available as static PDF files. Although these are retrospective reports, they include some commentary on hydrologically significant events such as groundwater droughts.

Knowledge gap 10: Although a near-real time UKCEH web portal is available to monitor and visualise water resource status in the UK (including groundwater resources), and monthly, retrospective commentaries are available on water resources in the UK, there is no specific resource or web-service that brings data, information and knowledge together in near real-time related to droughts (including groundwater droughts), i.e. England does not have an on-line drought monitor.

4. Modelling and forecasting groundwater droughts and their impacts

Compared with other types of hydrological drought, there is only a limited international peer-reviewed literature related to modelling of aspects of groundwater droughts and in particular a very limited literature specifically related to groundwater droughts in England (note, however, that there has been work by a number of water companies and the consultancy sector to develop models to better constrain groundwater availability during droughts, e.g. HR Wallingford (2020) but that work is out of scope of the current review). In addition, those modelling studies that have been published are diverse in both their research aims and the approaches and methods they adopted. So, it is difficult to identify consistent research themes or challenges related to groundwater modelling of droughts. Some examples of national- or larger-scale groundwater drought modelling studies include:

- Li and Rodell (2015) used Catchment Land Surface Models (CLSM) to estimate groundwater drought status in observation-poor areas at the continental scale. They generally obtained satisfactory results, however model parameters that control the depth to the water table, including bedrock depth, were shown to strongly influence the evolution and persistence of modelled groundwater droughts.
- Hellwig et al. (2020) employed a high-resolution transient groundwater model to investigate groundwater drought dynamics across Germany. Model performance was poorer in mountain regions where resolution was too low to represent local valley aquifers. They found that optimal precipitation accumulation times varied from few months in the Central German Uplands to several years in the porous aquifers in northern Germany and that in turn corresponded with regionally distinct groundwater drought dynamics.
- Schuler et al. (2022) used machine learning techniques (Random Forests regressor) to model groundwater memory at 114 sites across Ireland as a precursor to developing a groundwater drought susceptibility assessment for the country. They found relative and absolute topography and overburden thickness were the key explanatory variables of groundwater memory.
- There are multiple challenges in providing timely groundwater drought resource status mappings and forecasts at an appropriate spatial resolution to reflect the heterogeneous response of groundwater systems to continually changing climate drivers. To address this challenge, Brakkee et al. (2022) tested the use of impulse-response time series models to reconstruct the spatio-temporal development of the 2018–2019 groundwater drought in the south-eastern Netherlands. They found that the time series modelling was a useful tool to reconstruct regional groundwater drought development, however, they noted that “the use of time series simulations rather than direct measurement series [for groundwater drought monitoring] can bias drought estimations, especially at a local scale, and underestimate spatial variability”.

Given the significance of drought propagation through the different components of the terrestrial water cycle, it is expected that there will be an increasing need to develop integrated surface water-groundwater models of water cycle dynamics under drought. A recent example of such a linked groundwater-surface water model of drought dynamics is that of Kang and Sridhar (2019) who modelled droughts in the Chesapeake Bay catchment, USA. Kang and Sridhar (2019) coupled a VIC model with MODFLOW and assessed a range of model structures to investigate which was most effective at modelling a Multivariate Standardized Drought Index (MSDI) and concluded that “the coupled framework including surface and groundwater conditions is useful for considering surface and groundwater dynamics while assessing the impact of changing hydrology on drought predictions”. Mortazavi-Naeini et al. (2019) used a coupled modelling system that combined hydrological models of streamflow and quality and abstraction to assess the impact of climate change, change in demand, and land-use change on the reliability of public water supplies in the Thames Basin under low flows. However, as Ascott et al. (2021) noted, groundwater was represented very simplistically in the modelling scheme used by Mortazavi-Naeini et al. (2019) and Ascott et al. (2021) identified the need to ensure appropriate representation of groundwater process in future integrated models of low flows and drought. Recent research that supports this conclusion is the unpublished PhD study of Rodriguez-Yebra (2020). This study undertook detailed modelling of the drought response of an adited Chalk groundwater supply source adjacent to a river in England where there were good local observations; MODFLOW-USG and the Connected Linear Network package were used to simulate the aquifer and the source's boreholes and adits. It was demonstrated that river-aquifer connectivity varied greatly during the development and period of termination of the 2010-2012 drought and that the drought response could not be reproduced using existing conceptual models of river-aquifer interaction within the MODFLOW code. A new conceptual model of river aquifer interaction at the site was developed, which has implications for the understanding of the timing of the impact of abstraction on river flows. The study highlighted the lack of knowledge about hydrogeological and river-aquifer interaction processes operating during drought because of the limited availability observational data at the required spatio-temporal scales.

Knowledge gap 11: Modelling of groundwater droughts in England (and more widely) is immature compared with that of floods. A variety of methods are potentially available from stochastic and data-led, lumped parameter and distributed groundwater models, to machine learning approaches. However, the respective utility and applicability of such models to a range of groundwater drought-related research and management questions is unclear. In addition, coupled models of water resources, drought, climate change and water supply been developed for research purposes, and these invariably represent groundwater in the most simplistic manner. There is a need for a thorough review of modelling requirements related to groundwater-aspects of drought in England with particular emphasis on the most appropriate approaches to integrate groundwater processes into linked models of drought dynamics, and the observational data needed to evaluate their skill.

In the UK a number of groundwater-specific modelling studies have reconstructed or forecast groundwater level or SGI time series at monthly to centennial time scales. Although not the main focus of the studies, they are potentially of significance in enabling changes in groundwater drought frequency and characteristics over time to be investigated. However, to date, analyses of their outputs have not been published in this context. For example:

- Jackson et al. (2016) described a lumped-parameter groundwater model to reconstruct groundwater levels at observations boreholes and illustrated it by applying it to six sites in England: Skirwith, Swan House, New Red Lion, Bussels No. 7a, Chilgrove House, and Lower Barn Cottage. Groundwater levels were reconstructed from 2012 back to 1910 so making the hydrographs potentially amenable to and analysis of droughts and their changing characteristics over that period (although that analysis was not undertaken).
- Jackson et al. (2015) summarised the work of the Future Flows project (Prudhomme et al., 2012) where lumped parameter models for a number of hydrographs at observation boreholes across England used UK Met Office's UKCP09 probabilistic climate predictions to forecast monthly groundwater levels out to the 2050s.

- The work of (Prudhomme et al., 2012) has recently been updated as part of the eFLaG project with model forecasts for 54 observations boreholes (Hannaford et al., 2022) and a national groundwater recharge model for the UK (Hughes et al, 2021) driven by UK Met Office's UKCP18 out to the 2080s. An analysis of droughts in the river flow and groundwater level projections from the eFLaG project has been undertaken and submitted for publication (Parry et al, 2023.); this has indicated some differences between changes in projected low river flows and groundwater levels to future climate due to differences in the response of surface water and groundwater systems to summer and winter rainfall, which has important implications for water resources management planning.
- As previously mentioned in section 3, as part of a study to investigate historical episodes of drought, Bloomfield et al. (2018b; 2018c) used the methodology of Jackson et al. (2016) to reconstruct groundwater level hydrographs for 54 observation boreholes across the UK and reported the data as levels and SGI time series. However, groundwater droughts have not been extracted from the reconstructions.

Given the infrequency of major episodes of groundwater drought, there would be benefit in bringing together reconstructed data, observational data and forecasted data for groundwater level and SGI time series at observation boreholes across England to assess the evidence for changes in groundwater drought frequency and characteristics over the period covered, i.e. for almost two centuries from the 1890s to 2080s. This analysis would of course need to take into account the associated uncertainties with the reconstructed and forecast data.

Knowledge gap 12: Despite almost two centuries of reconstructed, observed and modelled groundwater level and SGI time series being available for over 50 observation boreholes across the UK, a globally unique data resource, there has been no attempt to date to extract and analyse groundwater droughts from long observed/modelled records and no assessment of the evidence they provide in constraining changes in groundwater drought frequency and characteristics over this period.

Wendt et al. (2021) recently reported on a socio-hydrological modelling study of the effects of a range of water resource management strategies on groundwater drought outcomes using idealised virtual catchments based on climate data, water resource management practices and drought policies representative of England. A water balance model was used to test four scenarios separately and in combination, as follows: increased water supply, restricted water demand, conjunctive use, and maintenance of environmental flows through restrictions to groundwater abstractions. They demonstrated "mitigated droughts for both baseflow and groundwater droughts in scenarios applying conjunctive use, particularly in systems with small groundwater storage" and "low sensitivity of … drought management strategies to different hydrogeological conditions". It is clear from this study and other work (Ascott et al, 2021) that more needs to be done to understand the impacts of water management practices on groundwater drought propagation. One factor holding such modelling studies back is the challenge of obtaining information on abstractions and discharges for real catchments (Ascott et al, 2021). Some long-term, catchment-averaged information is now available for selected water management practices (abstractions, discharges and reservoir storage) in the CAMELS-GB open-source dataset (Coxon et al., 2020a; 2020b), however this is the exception and much more needs to be done to make such information more freely available.

Knowledge gap 13: Despite evidence for the importance of including water resource management data in models of groundwater drought dynamics, there is limited freely available information on abstractions and discharges from and to managed groundwater systems or information on specific water resource management responses during episodes of drought other than that published in water company Drought Plans.

5. Summary of research gaps and opportunities and priorities for future research

There have been a couple of recent opinion pieces in the peer-reviewed literature relevant to research needs related to groundwater droughts in England (Ascott et al., 2021; Wagener et al., 2021).

Ascott et al. (2021) recently reported on a day-long workshop as part of the NERC-funded Drought and Water Scarcity Programme considering the current status and future priorities for research on groundwater supplies during droughts in England. They identified four primary needs, as follows:

1. integration of definitions of drought,
2. enhanced fundamental monitoring of drought stressed groundwater systems,
3. integrated modelling of groundwater in the water cycle, and
4. better information sharing.

As noted in the Introduction, this essay has focussed on groundwater aspects of drought and Knowledge gaps 1 to 3 (tabulated below) relate to the first Ascott et al. (2021) primary research needs, i.e. to drought definition. Ascott et al. (2021) call for integration of drought definitions across all aspects of the terrestrial water cycle and that view is endorsed here. Ascott et al. (2021) call for enhanced fundamental monitoring of drought stressed groundwater systems. This call is also endorsed here in Knowledge gap 4, with a particular emphasis on the requirement for joined-up monitoring across the terrestrial water cycle from the monitoring of precipitation, soil moisture and discharge to groundwater storge, discharge and stream flow in common catchments. The need for integrated modelling of groundwater in the water cycle (Ascott et al., 2021) is also identified here (Knowledge gap 11). Ascott et al. (2021) also called for better information sharing. This is also identified here as Knowledge gap 13 with a particular emphasis on the need for improved access to data related water resource management practices for inclusion in models of groundwater drought dynamics.

Wagener et al. (2021) identified a series of knowledge gaps that they believe need to be addressed if we are to develop a coherent perceptual model of the hydrology (including hydrogeology) of Great Britain. The first of these was to be able to account for groundwater fluxes to close open water balances in regions of highly permeable aquifers – a particular challenge under conditions of groundwater drought. They highlighted the need for "nationally consistent perceptual understanding of catchment and aquifer controls on spatio-temporal variation in recharge and groundwater discharge to rivers" – this relates closely to Knowledge gap 4 listed below. They also highlighted the need for a better understanding of climate change impacts on hydrology across Great Britain and particularly emphasised the importance of a good perceptual model of the role groundwater in drought propagation and the sensitivity of aquifer such as the Chalk to groundwater drought, asking questions such as "which catchments will see the drought signal move through soil moisture and groundwater stores more quickly than others, and which catchments will recover first when the drought subsides?". These research challenges map directly onto Knowledge gaps 4, 5, 8, 9, and 12 below. To help address these and other research challenges that they identified they emphasised the need for improved access to hydrological data and particularly data related to "soils, on land cover, on groundwater, and on human activities (especially abstractions and reservoir management)", mirroring our Knowledge gap 13 below.

The knowledge or research gaps identified in this essay are tabulated below. For each knowledge or research gap, opportunities for future research have been identified and given a relative priority.

Knowledge or research gap	Opportunity for future research	Priority
Knowledge gap 1: Which, if any, EA regulatory functions or processes require or would benefit from a formal definition of groundwater droughts? Is there a need or benefit in discriminating between groundwater recharge, storage and discharge droughts in different regulatory settings? How would such formal definitions of groundwater drought relate to wider formal definitions of drought that may also be used in a regulatory context?	Review which EA regulatory functions use the concept of 'groundwater drought' and identify if it is possible and desirable to discriminate between groundwater recharge, storage and discharge droughts in different regulatory settings.	H
Knowledge gap 2: No guidance is currently available for EA staff and their contractors regarding the different methods to characterise and quantify groundwater droughts, including the pros and cons of the	Commission a systematic review of methods used to define meteorological, hydrological and other droughts. To include pros	H

different methods, the implications for the nature of any groundwater droughts so defined, and the suitability of a given method for specific EA regulatory procedures or requirements.	and cons of each method and implications for droughts defined.	
Knowledge gap 3: Within water company drought management plans in England, there is no consistent approach to groundwater drought quantification and characterisation, and in particular to the setting of drought trigger levels. Consequently, it is not possible to assess if water companies are using consistent trigger levels in response to an episode of drought. What are the implications of this for drought management planning; what would be the benefits, if any, of encouraging the use of consistent definitions of groundwater droughts across water company Drought Management Plans; and, what would be the most suitable approaches to drought definition, e.g. use of SGI?	Review drought definitions in water company Drought Management Plans. Scope of review to include documentation and assessment of the methods used, suitability for meeting regulatory needs, recommendations for consistent methodology.	L
Knowledge gap 4: There is no high storage catchment in England with long, co-located observational records of driving meteorology, soil moisture, groundwater recharge, storage and discharge, and surface flows to investigate processes associated with drought initiation, propagation through the terrestrial water cycle, and termination in a groundwater-dominated catchment.	Establish, in catchments representative of the range of hydrogeological settings in the UK, instrumentation to characterise drought propagation through the full terrestrial water cycle. Note that this is an explicit aim for the new UKRI/NERC-funded Floods and Droughts Research Infrastructure (FDRI) initiative (UKCEH, 2023c).	H
Knowledge gap 5: It is evident from several studies that groundwater	Map the sensitivity of groundwater systems in England to drought.	H

response to large meteorological droughts is spatially variable, particularly in the Chalk aquifer, and that there is spatial variation in the sensitivity of groundwater to meteorological droughts of different durations and magnitudes. However, unlike groundwater flooding, there is no mapping of the spatial variation in groundwater susceptibility or sensitivity to drought across England. There is currently no national volumetric assessment of Chalk storage and the sensitivity of storage to drought.	Using observational data, quantify the spatio-temporal relationships between SPI and SGI (i.e. significant SPI accumulation periods, SPI-SGI cross-correlations, and lags in drought propagation and termination) and map these across England. (see also Knowledge gaps 8 and 12). Produce an updated Chalk storage model for England (based on the approach outlined in NRA, 1993) to address the question “how much water is left” and act as a management and communication tool during future droughts	
Knowledge gap 6: There is no typology for groundwater drought impacts for the UK (or internationally) and no institution or group is actively collating and maintaining a record of groundwater drought impacts in an inventory either for regulatory or research purpose. This is an essential precursor to any future quantitative assessment of the links between driving climatology, groundwater drought status and impacts of groundwater drought.	Establish a typology for groundwater drought impacts for the UK as a precursor to developing an inventory of groundwater drought impacts. The inventory should be populated, where possible with the available historical data and maintained to record future impacts. It should be a part of a wider drought impact inventory for England with a consistent typology.	H
Knowledge gap 7: There is generally a poor understanding of borehole yields during episodes of major droughts.	Undertake step tests for a range of abstraction borehole in a variety of hydrogeological contexts under extreme drought conditions.	L
Knowledge gap 8: Despite the available data, there has been no systematic longitudinal overview and analysis of groundwater droughts in the observed record for England and how they related to the driving climatology. This is a significant omission given the importance of	See Knowledge gap 12.	H

understanding and developing 'benchmark groundwater droughts' for reference for planning responses to future drought.		
Knowledge gap 9: There is evidence that a number of decadal-scale changes influence the occurrence and nature of droughts in England. These include decadal-scale changes in atmospheric circulation, anthropogenic climate change, and changes in water resource management practices in England. However, to date there has been no work to synthesise this evidence, to investigate the relative magnitude of these effects, to investigate if these effects are compounding (for example are they additive or multiplicative?), and if and how understanding of the change-drivers of groundwater drought should inform future modelling, forecasting, planning and operational management activities related to groundwater droughts.	Undertake a review of the relative importance, and interaction of the multiple drivers of change in groundwater dynamics at the decadal time scale and asses the implications for future modelling, forecasting, planning and operational management activities related to groundwater droughts.	M
Knowledge gap 10: Although a near-real time UKCEH web portal is available to monitor and visualise water resource status in the UK (including groundwater resources), and monthly, retrospective commentaries are available on water resources in the UK, there is no specific resource or web-service that brings data, information and knowledge together in near real-time related to droughts (including groundwater droughts), i.e. England does not have an on-line drought monitor.	Establish a near real-time web portal that delivers groundwater drought status and associated information as part of a comprehensive drought monitoring service for England	M
Knowledge gap 11: Modelling of groundwater droughts in England (and	Review modelling approaches that enable groundwater processes to	M

more widely) is immature compared with that of floods. A variety of methods are potentially available from stochastic and data-led, lumped parameter and distributed groundwater models, to machine learning approaches. However, the respective utility and applicability of such models to a range of groundwater drought-related research and management questions is unclear. In addition, coupled models of water resources, drought, climate change and water supply been developed for research purposes, and these invariably represent groundwater in the most simplistic manner. There is a need for a thorough review of modelling requirements related to groundwater-aspects of drought in England with particular emphasis on the most appropriate approaches to integrate groundwater processes into linked models of drought dynamics, and the observational data needed to evaluate their skill.	be incorporated in water resource modelling systems and identify best practice for coupled models applied to the estimation and prediction of low flows and droughts. The review should include an assessment of current operational water resource management models that have an application in the context of droughts, the extent to which those models currently incorporate groundwater information, and recommendations for the improved representation of groundwater systems in such models.	
Knowledge gap 12: Despite almost two centuries of reconstructed, observed and modelled groundwater level and SGI time series being available for over 50 observation boreholes across the UK, a globally unique data resource, there has been no attempt to date to extract and analyse groundwater droughts from long observed/modelled records and no assessment of the evidence they provide in constraining changes in groundwater drought frequency and characteristics over this period.	(see also Knowledge gap 8) Combine groundwater level and SGI data from representative observation boreholes with published reconstructed levels and SGI time series (from Jackson et al, 2016) Bloomfield et al. (2018b; 2018c) and with forecasts from groundwater levels from the eFLaG project. Taking into account model uncertainties, define and extract groundwater droughts in the combined modelled / observed time series. Characterise the extracted droughts in terms of their frequency/return period, drought characteristics (e.g. magnitude,	H

	intensity and duration), and how they vary spatially and by aquifer type. Investigate the spatio-temporal relationships between the extracted groundwater drought and the driving climatology and how this has and will change in the future.	
Knowledge gap 13: Despite evidence for the importance of including water resource management data in models of groundwater drought dynamics, there is limited freely available information on abstractions and discharges from and to managed groundwater systems or information on specific water resource management responses during episodes of drought other than that published in water company Drought Plans.	Water companies, environmental regulators and research community work together to make water resource management data more freely available for the purposes of improving groundwater drought forecasts more accurate and reliable and to improve the effectiveness of drought planning process.	M

References

ASCOTT, M.J., MANSOUR, M., BLOOMFIELD, J.P. & UPTON, K.A. 2019. Analysis of the impact of hydraulic properties and climate change on estimations of borehole yields. Journal of Hydrology 577, 123998. https://doi.org/10.1016/j.jhydrol.2019.123998

ASCOTT, M.J.; BLOOMFIELD, J.P ; KARAPANOS, I.; JACKSON, C.R.; WARD, R.S.; MCBRIDE, A.B.; DOBSON, B.; KIEBOOM, N.; HOLMAN, I.P.; VAN LOON, A.F.; CRANE, E.J.; BRAUNS, B.; RODRIGUES-YEBRA, A.; UPTON, K.A. 2021. Managing groundwater supplies subject to drought: perspectives on current status and future priorities from England (UK). Hydrogeology Journal 29, 921-924. https://doi.org/10.1007/s10040-020-02249-0

BARTHEL, R., STANGEFELT, M., GIESE, M., NYGREN, M., SEFTIGEN K. & CHEN D. 2021. Current understanding of groundwater recharge and groundwater drought in Sweden compared to countries with similar geology and climate, Geografiska Annaler: Series A, Physical Geography 103:4, 323-345, doi: 10.1080/04353676.2021.1969130

BEYENE, B. S. & VAN LOON, A. F 2014. Investigation of variable threshold level approaches for hydrological drought identification, Hydrol. Earth Syst. Sci. Discuss 12765–12797, doi:10.5194/hessd-11-12765-2014.

BEESON, S., MISSTEAR, B.D. & VAN WONDEREN, J.J.1997. Assessing the reliable outputs of groundwater sources. Water Environ. J. 11 (4), 295–304. https://doi.org/10.1111/j.1747-6593.1997.tb00132.x.

BHUIYAN, C., SINGH, R.P. & KOGAN, F.N 2006. Monitoring drought dynamics in the Aravalli region (India) using different indices based on ground and remote sensing data. Int. J. Appl. Earth Observ. Geoinf. 8, 289–302.

BLOOMFIELD J P & MARCHANT B P 2013. Analysis of groundwater drought building on the standardised precipitation index approach. Hydrol Earth Syst Sci, 17:4769–4787.

BLOOMFIELD J P, MARCHANT B P, BRICKER S H & MORGAN R B 2015. Regional analysis of groundwater droughts using hydrograph classification. Hydrology and Earth System Sciences 19, 4327-4344

BLOOMFIELD J P, ASCOTT, M., MARCHANT, B., MACDONALD, D. & MCKENZIE, A. 2018a. Modelling asymmetries in groundwater drought and flood status across the Chalk aquifer of the UK since 1960. EGU, Vienna, 13th April 2018, session HS2.1.1 - Hydrological extremes: from droughts to floods. Geophysical Research Abstracts, Vol. 20, EGU2018-4519, 2018 https://meetingorganizer.copernicus.org/EGU2018/EGU2018-4519.pdf

BLOOMFIELD, J.P.; MARCHANT, B.P. & WANG, L. 2018b. Historic reconstructions of monthly groundwater levels for 54 UK boreholes (1891-2015). NERC Environmental Information Data Centre. https://doi.org/10.5285/ccfded8f-c8dc-4a24-8338-5af94dbfcc16

BLOOMFIELD, J.P.; MARCHANT, B.P. & WANG, L 2018c. Historic Standardised Groundwater level Index (SGI) for 54 UK boreholes (1891-2015). NERC Environmental Information Data Centre. https://doi.org/10.5285/d92c91ec-2f96-4ab2-8549-37d520dbd5fc

BLOOMFIELD J P, MARCHANT B J & MCKENZIE A A. 2019. Changes in groundwater drought associated with anthropogenic warming. Hydrol. Earth Syst. Sci. 23, 1393–1408

BRAKKEE, E., VAN HUIJGEVOORT, M.H.J. & BARTHOLOMEUS, R.P. 2022. Improved understanding of regional groundwater drought development through time series modelling: the 2018–2019 drought in the Netherlands. Hydrol. Earth Syst. Sci. 26, 551–569, 2022

CHANG, T.J. & TEOH, C.B. 1995. Use of the Kriging method for studying characteristics of ground water droughts. J. Am. Water Resour. Assoc. 257, 1001–1007

CHANGNON JR SA 1987. Detecting drought conditions in Illinois. Illinois State Water Survey Champaign, Circular 169.

COXON, G.; ADDOR, N.; BLOOMFIELD, J.P.; FREER, J.; FRY, M.; HANNAFORD, J.; HOWDEN, N.J.K.; LANE, R.; LEWIS, M.; ROBINSON, E.L.; WAGENER, T. & WOODS, R 2020a. Catchment attributes and hydro-meteorological timeseries for 671 catchments across Great Britain (CAMELS-GB). NERC Environmental Information Data Centre. https://doi.org/10.5285/8344e4f3-d2ea-44f5-8afa-86d2987543a9

COXON, G., ADDOR, N., BLOOMFIELD, J. P., FREER, J., FRY, M., HANNAFORD, J., HOWDEN, N. J. K., LANE, R., LEWIS, M., ROBINSON, E. L., WAGENER, T. & WOODS, R 2020b. CAMELS-GB: hydrometeorological time series and landscape attributes for 671 catchments in Great Britain, Earth Syst. Sci. Data, 12, 2459–2483, https://doi.org/10.5194/essd-12-2459-2020

ELTAHIR, E.A.B. & YEH, P.J.F. 1999. On the asymmetric response of aquifer water level to floods and droughts in Illinois. Water Resour. Res. 35 (4), 1199–1217

EUROPEAN DROUGHT CENTRE 2022. European Drought Impact Report Inventory (EDII Database), https://www.geo.uio.no/edc/droughtdb/edr/impactdatabase.php

ENVIRONMENT AGENCY 2022. Water Situation Reports. https://www.gov.uk/government/collections/water-situation-reports-for-england

ELTAHIR, E.A.B. & YEH, P.J.F. 1999. On the asymmetric response of aquifer water level to floods and droughts in Illinois. Water Resour. Res. 35 (4), 1199–1217.

FENDEKOVA M AND FENDEK M. 2012. Groundwater drought in the Nitra River Basin – Identification and classification. J. Hydrol. Hydromech. 60, 3, 185–193

FIORILLO F & GUADAGNO F M 2010. Karst spring discharges analysis in relation to drought periods, using the SPI. Water Resour Manag 24,1864–1884

FIORILLO F & GUADAGNO F M 2012. Long karst spring discharge time series and droughts occurrence in Southern Italy. Environ Earth Sci 65, 2273–2283, doi: 10.1007/s12665-011-1495-9

FOLLAND C K., HANNAFORD J, BLOOMFIELD J P., KENDON M., SVENSSON C, MARCHANT B P., PRIOR J & WALLACE. 2015. Multi-annual droughts in the English Lowlands: a review of their characteristics and climate drivers in the winter half-year. Hydrol. Earth Syst. Sci. 19, 2353–2375.

GUO M, YUE W, WANG T, ZHANG N & WU L. 2021. Assessing the use of standardized groundwater index for quantifying groundwater drought over the conterminous US. Journal of Hydrology 598, 126227

HANNAFORD J, LLOYD-HUGHES B, KEEF C, PARRY S & PRUDHOMME C. 2011. Examining the large-scale spatial coherence of European drought using regional indicators of precipitation and streamflow deficit. Hydrol Process 25,1146–1162. doi:10.1002/hyp.7725.

HANNAFORD, J., MACKAY, J., ASCOTT, M., BELL, V., CHITSON, T., COLE, S., COUNSELL, C., DURANT, M., JACKSON, C. R., KAY, A., LANE, R., MANSOUR, M., MOORE, R., PARRY, S., RUDD, A., SIMPSON, M., FACER-CHILDS, K., TURNER, S., WALLBANK, J., WELLS, S., & WILCOX, A. 2022. eFLaG: enhanced future FLows and Groundwater. A national dataset of hydrological projections based on UKCP18, Earth Syst. Sci. Data Discuss. [preprint], https://doi.org/10.5194/essd-2022-40

HELLWIG, J & STAHL, K 2018. An assessment of trends and potential future changes in groundwater-baseflow drought based on catchment response times, Hydrol. Earth Syst. Sci. 22, 6209–6224, https://doi.org/10.5194/hess-22-6209-2018

HELLWIG, J., DE GRAAF, I. E. M., WEILER, M., & STAHL, K. 2020. Large-scale assessment of delayed groundwater responses to drought. Water Resources Research 56, e2019WR025441. https://doi.org/10.1029/2019WR025441

HELLWIG, J., STOELZLE, M. & STAHL. K. 2021. Groundwater and baseflow drought responses to synthetic recharge stress tests. Hydrol. Earth Syst. Sci. 25, 1053–1068, 2021. https://doi.org/10.5194/hess-25-1053-2021

HEUDORFER, B & STAHL, K 2017. Comparison of different threshold level methods for drought propagation analysis in Germany. Hydrol. Res. 48, 1311–1326, https://doi.org/10.2166/nh.2016.258, 2017.

HR WALLINGFORD 2020. Updated projections of future water availability for the third UK. Climate Change Risk Assessment Technical Report MAR6025-RT002-R05-00, July 2020 https://www.ukclimaterisk.org/wp-content/uploads/2020/07/Updated-projections-of-future-water-availability_HRW.pdf

HUGHES, A.; MANSOUR, M.; WARD, R.; KIEBOOM, N.; ALLEN, S.; SECCOMBE, D.; CHARLTON, M. & PRUDHOMME, C. 2021 The impact of climate change on groundwater recharge: National-scale assessment for the British mainland. Journal of Hydrology 598, 126336. https://doi.org/10.1016/j.jhydrol.2021.126336

JACKSON C R, BLOOMFIELD J P & MACKAY J D. 2015. Evidence for changes in historic and future groundwater levels in the UK. Progress in Physical Geography 39(1), 49-67.

JACKSON C R, WANG L, PACHOCKA M, MACKAY J D & BLOOMFIELD J P 2016. Reconstruction of multi-decadal groundwater level time-series using a lumped conceptual model. Hydrol. Process. 30, 3107–3125

KENDON M, MARSH T & PARRY S 2013. The 2010-2012 drought in England and Wales. Water 68, 99-112

KUMAR R, MUSUUZA J L, VAN LOON A F, TEULING A J, BARTHEL R, TEN BROEK J, MAI J, SAMANIEGO L & ATTINGER S 2016. Multiscale evaluation of the standardized precipitation index as a groundwater drought indicator. Hydrol. Earth Syst. Sci. 20 1117–31

LANGE, BETTINA; HOLMAN, IAN & BLOOMFIELD, JOHN P 2016 A framework for a joint hydro-meteorological-social analysis of drought. Science of the Total Environment 578, 297-306. https://doi.org/10.1016/j.scitotenv.2016.10.145

LI, B. AND RODELL, M., 2015. Evaluation of a model-based groundwater drought indicator in the conterminous US. Journal of Hydrology, 526, pp.78-88.

MARCHANT, B.P. & BLOOMFIELD, J.P. 2018. Spatio-temporal modelling of the status of groundwater droughts. Journal of Hydrology 564, 397-413

MARCHANT, B.P., CUBA, D., BRAUNS, B. & BLOOMFIELD, J.P. 2022. Temporal interpolation of groundwater level hydrographs for regional drought analysis using mixed models. Hydrogeology Journal 30,1801–1817

MACKAY, J.D., JACKSON, C.R., BROOKSHAW, A., SCAIFE, A.A,; COOK, J. & WARD, R.S. 2015 Seasonal forecasting of groundwater levels in principal aquifers of the United Kingdom. Journal of Hydrology 530. 815-828.

MARSH, T. & PARRY, S. 2012. An overview of the 2010-12 drought and its dramatic termination. Wallingford, UK, NERC/Centre for Ecology & Hydrology, 4pp. (CEH Project Number: C04215) https://nora.nerc.ac.uk/id/eprint/20083/

MCKEE, T.B, DOESKEN, N.J.& KLEIST, J. 1993. The relationship of drought frequency and duration to time scales. In Eight Conference on Applied Climatology, 17–22 January 1993, Anaheim, California, Available at: http://ccc.atmos.colostate.edu/relationshipofdroughtfrequency.pdf

MENDICINO, G., SENATORE, A. & VERSACE, P. 2008. A Groundwater Resource Index (GRI) from drought monitoring and forecasting in a Mediterranean climate, J. Hydrol. 357, 282–302.

MISHRA, A. K. & SINGH, V. P 2010. A review of drought concepts. J. Hydrol. 391(1–2), 202–216.

MISSTEAR, B.D.R. & BEESON, S. 2000. Using operational data to estimate the reliable yields of water-supply wells. Hydrogeol. J. 8 (2), 177–187. https://doi.org/10.1007/s100400050004

MORTAZAVI-NAEINI, M., BUSSI, G., ELLIOTT, J. A., HALL, J. W., & WHITEHEAD, P. G. 2019. Assessment of risks to public water supply from low flows and harmful water quality in a changing climate. Water Resources Research 55, 10,386-10,404, https://doi.org/10.1029/2018WR022865

NATIONAL RIVER AUTHORITY. 1993. Groundwater storage in British aquifers: Chalk. National Rivers Authority, Project record 128/8/A

ORIMOLOYE, I.R. 2022. Agricultural Drought and Its Potential Impacts: Enabling Decision-Support for Food Security in Vulnerable Regions. Front. Sustain. Food Syst. 6, 838824. doi: 10.3389/fsufs.2022.838824

PARRY, S., PRUDHOMME, C., WILBY, R.L. & PAUL J. WOOD. 2016. Drought termination: Concept and characterisation. Progress in Physical Geography 40 (6) 743–767

PARRY, S., PRUDHOMME, C., WILBY, R.L., PAUL J. WOOD & MCKENZIE, A. 2018. Demonstrating the utility of a drought termination framework: prospects for groundwater level recovery in England and Wales in 2018 or beyond. Environ. Res. Lett. 13, 064040 .

PARRY, S., MACKAY, J. D., CHITSON, T., HANNAFORD, J., BELL, V. A., FACER-CHILDS, K., KAY, A., LANE, R., MOORE, R. J., TURNER, S., AND WALLBANK, J. 2023. Divergent future drought projections in UK river flows and groundwater levels, Hydrol. Earth Syst. Sci. Discuss. [preprint], https://doi.org/10.5194/hess-2023-59, in review, 2023.

PETERS, E., TORFS, P.J.J.F, VAN LANEN, H.A.J. & BIER, G. 2003. Propagation of drought through groundwater - a new approach using linear reservoir theory. Hydrol Process 17:3023–3040. doi:10.1002/hyp.1274.

PETERS, E., VAN LANEN, H.A.J., TORFS, P.J.J.F. & BIER, G. 2005. Drought in groundwater—drought distribution and performance indicators. Journal of Hydrology 306, 302–317

PETERS, E., BIER, G., VAN LANEN ,H.A.J. & TORFS, P.J.J.F.2006. Propagation and spatial distribution of drought in a groundwater catchment. J Hydrol. 321,257–275. doi:10.1016/j.jhydrol.2005.08.004.

PETERSEN-PERLMAN, J.D., AGUILAR-BARAJAS, I. & MEGDAL, S.B. 2022. Drought and groundwater management: Interconnections, challenges, and policy responses. Current Opinion in Environmental Science & Health, 28,100364

PRUDHOMME C, CROOKS S, JACKSON CR, ET AL. 2012. Future flows and groundwater levels: Final technical report. Report SC090016/PN9. Wallingford: Centre for Ecology & Hydrology.

RODRIGUEZ-YEBRA, A. 2020. Modelling the drought response of adited Chalk groundwater sources. Unpublished PhD thesis, Imperial College London. https://doi.org/10.25560/100521

RUST R, HOLMAN I, CORSTANJE R, BLOOMFIELD J P & CUTHBERT M 2018. A conceptual model for climatic teleconnection signal control on groundwater variability in the UK and Europe. Earth-Science Reviews 177, 164-174

RUST W, HOLMAN I, BLOOMFIELD J P, CUTHBERT M & CORSTANJE R 2019. Understanding the potential of climate teleconnections to project future groundwater drought. Hydrol. Earth Syst. Sci. 23, 3233–3245

RUST W, CUTHBERT M, BLOOMFIELD J P, CORSTANJE R, HOWDEN N & HOLMAN I 2021. Exploring the role of hydrological pathways in modulating North Atlantic Oscillation (NAO) teleconnection periodicities from UK rainfall to streamflow. Hydrol. Earth Syst. Sci. 25, 2223–2237.

RUST W, BLOOMFIELD J P, CUTHBERT M, CORSTANJE R &HOLMAN I 2022. The importance of non-stationary multiannual periodicities in the NAO index for forecasting water resource extremes. Hydrol. Earth Syst. Sci. 26, 2449–2467

SCHREINER-MCGRAW, A.P. & AJAMI, H. 2021. Delayed response of groundwater to multi-year meteorological droughts in the absence of anthropogenic management. Journal of Hydrology 603, 126917

SCHULER, P., CAMPANYA, J., MOE, H., DOHERTY, D., HUNTER WILLIAMS, N. & MCCORMACK, T. 2022. Mapping the groundwater memory across Ireland: A step towards a groundwater drought susceptibility assessment. Journal of Hydrology 612, 128277

SLETTE I J, POST A K, AWAD M, ET AL. 2019 How ecologists define drought, and why we should do better. Glob. Change Biol. 25, 3193–3200. https://doi.org/10.1111/gcb.14747

STAGGE J H, TALLAKSEN L M, GUDMUNDSSON L, VAN LOON A F & STAHL K 2015a. Candidate Distributions for Climatological Drought Indices (SPI and SPEI). Int. J. Climatol. 35, 4027–4040.

STAGGE, J.H., KOHN, I., TALLAKSEN, L.M. & STAHL, K. 2015b. Modeling drought impact occurrence based on meteorological drought indices in Europe. Journal of Hydrology 530, 37–50

STAHL, K., KOHN, I., BLAUHUT, V., URQUIJO, J., DE STEFANO, L., ACÁCIO, V., DIAS, S., STAGGE, J. H., TALLAKSEN, L.M., KAMPRAGOU, E., VAN LOON, A.F., BARKER, L.J., MELSEN, L.A., BIFULCO, C., MUSOLINO, D., DE CARLI, A., MASSARUTTO, A., ASSIMACOPOULOS, D., VAN LANEN, H.A.J. 2015. Impacts of European drought events: insights from an international database of text-based reports. Nat. Hazards Earth Syst. Sci. 16, 801–819.

TALLAKSEN L M & VAN LANEN H A J 2004. Hydrological drought: processes and estimation methods for streamflow and groundwater. In: Developments in Water Science Vol. 48. Amsterdam, the Netherlands: Elsevier Science B.V.

TALLAKSEN L M, HISDAL H, VAN LANEN H A J 2009. Space-time modelling of catchment scale drought characteristics. J Hydrol 375, 363–372. doi:10.1016/j.jhydrol.2009.06.032.

THOMAS, B.F., FAMIGLIETTI, J.S., LANDERER, F.W., WIESE, D.N., MOLOTCH, N.P. & ARGUS, D.F. 2017. GRACE Groundwater Drought Index: Evaluation of California Central Valley groundwater drought. Remote Sensing of Environment 198, 384-392

UKCEH 2023a. Monthly Hydrological Summaries. https://nrfa.ceh.ac.uk/monthly-hydrological-summary-uk

UKCEH 2023b. Historic Droughts project. https://www.ceh.ac.uk/our-science/projects/historic-droughts

UKCEH 2023c. Floods and Droughts Research Infrastructure (FDRI). https://www.ceh.ac.uk/our-science/projects/floods-and-droughts-research-infrastructure-fdri

VAN LANEN, H.A.J. & PETERS, E 2000. Definition, effects and assessment of groundwater droughts. In: Vogt, J.V., Somma, F. (Eds.), Drought and Drought Mitigation in Europe. Kluwer Academic Publishers, Dordrecht, pp. 49–61.

VAN LOON A F. 2015. Hydrological drought explained. WIREs Water 2015, 2:359–392. doi: 10.1002/wat2.1085

VAN LOON A.F., GLEESON, T., CLARK, J., VAN DIJK, A.I.J.M., STAHL, K., HANNAFORD, J., BALDASSARRE, G.D. , TEULING, A.J., TALLAKSEN, L.M., UIJLENHOET, R., HANNAH, D.M., SHEFFIELD, J., SVOBODA, M., VERBEIREN, B. , WAGENER, T. , RANGECROFT, S., WANDERS, N. & VAN LANEN, H.A.J. 2016. Drought in the Anthropocene Nature Geoscience volume 9, pages89–91 (2016)

VICENTE-SERRANO, S.M., BEGUERÍA, S. & LÓPEZ-MORENO. J.I. 2009. A multiscalar drought index sensitive to global warming: the standardized precipitation evapotranspiration index. J Clim, 23,1696–1718

VICENTE-SERRANO, S., LÓPEZ-MORENO, J., BEGUERÍA, S., LORENZO-LACRUZ, J., AZORIN-MOLINA, C. & MORÁN-TEJEDA, E. 2012. Accurate computation of a streamflow drought index. J Hydrol Eng, 17, 318–332.

WAGENER, T., DADSON, S.J., HANNAH, D.M., COXON, G., BEVEN, K., BLOOMFIELD, J.P., BUYTAERT, W., CLOKE, H., BATES, P., HOLDEN, J. AND PARRY, L., 2021. Knowledge gaps in our perceptual model of Great Britain's hydrology. Hydrological Processes, 35(7), p.e14288.

WATTS G, BATTARBEE R, BLOOMFIELD J P, CROSSMAN J, DACCACHE A, DURANCE I, ELLIOT J, GARNER G, HANNAFORD J, HANNAH D M, HESS T, JACKSON C R, KAY A L, KERNAN M, KNOX J, MACKAY J, MONTEITH D T, ORMEROD S, RANCE J, STUART M E, WADE A J, WADE S D, WEATHERHEAD K, WHITEHEAD P G & WILBY R L. 2015. Climate change and water in the UK – past changes and future prospects. Progress in Physical Geography 39(1), 6-28.

WENDT DE, VAN LOON A F, BLOOMFIELD J P & HANNAH D M 2020. Asymmetric impact of groundwater use on groundwater droughts. Hydrol. Earth Syst. Sci. 24, 4853–4868 https://doi.org/10.5194/hess-24-4853-2020

WENDT D E, BLOOMFIELD J P, VAN LOON A F, GARCIA M, HEUDORFER B, LARSEN J & HANNAH D M 2021. Evaluating integrated water management strategies to inform hydrological drought mitigation   Nat. Hazards Earth Syst. Sci. 21, 3113–3139. https://doi.org/10.5194/nhess-21-3113-2021

WILBY R L 2006. When and where might climate change be detectable in UK river flows? Geophysical Research Letters 33, L19407.

WORLD METEOROLOGICAL ORGANIZATION 2012. Standardized precipitation index user guide. M. Svoboda,M. Hayes and D. Wood (WMO-No. 1090), Geneva.

WOSSENYELEH, B. K., WORKU, K. A., VERBEIREN, B. & HUYSMANS, M 2021. Drought propagation and its impact on groundwater hydrology of wetlands: a case study on the Doode Bemde nature reserve (Belgium). Nat. Hazards Earth Syst. Sci. 21, 39–51, https://doi.org/10.5194/nhess-21-39-2021, 2021

WOSSENYELEH B K, KASA A S, VERBEIREN B & HUYSMANS M 2022. Drought propagation in the hydrological cycle in a semiarid region: a case study in the Bilate catchment, Ethiopia, Hydrogeology Journal 30(3), 751-766

ZAIDMAN M D, REES H G & YOUNG A R 2002. Spatio-temporal development of streamflow droughts in northwest Europe. Hydrol Earth Syst Sci, 6, 733–751.

E: Review of the state of research on drought: Water quality

Mike Bowes, Alex O’Brien, Michael Hutchins and Cedric Laize.

UK Centre for Ecology & Hydrology.

Overview

Extreme low flows and long dry periods are predicted to have major impacts on river and lake water quality, as transfers of nutrients and pollutants from the catchment land surface to the waterbody will be minimised, and the impact of point source inputs will tend to be magnified due to the lack of dilution in the river or low flushing rates in lakes. The long residence times and often increased water temperatures and nutrient concentrations can also result in high algal and bacterial biomass and rapid biogeochemical cycling in rivers and lakes, which have important impacts on water quality and dissolved oxygen concentrations.

In general, droughts are expected to increase conductivity, water temperature, biological activity/biomass and point-source pollutants such as soluble phosphorus, metals, pharmaceuticals and personal care products. However, pH, overnight dissolved oxygen concentration and suspended sediment concentrations are likely to decrease.

Most of the individual biogeochemical processes that will occur during drought periods are largely understood and well-studied. However, droughts can result in many complex and competing processes. For instance, the lack of rainfall reduces nutrient and pollutant transfer from the catchment surface to the waterbody, thereby reducing concentrations. Conversely, reduced river flow and rates of lake flushing can cause anoxia in the bed-sediment, which can release P and metals into the waterbody, increasing concentration. In rivers dominated by continuous point-source inputs, pollution and nutrient loads will tend to become more concentrated as flow declines. However, increasing phosphorus and metal concentration in the water column will affect the equilibrium with the bed-sediment, increasing the sequestration by the bed and thereby reducing dissolved concentrations. How these processes and feedback loops interact is currently unknown.

Other competing relationships can potentially result in sudden step changes in water quality. For example, the relative proportion of groundwater to rivers and lakes will increase as the drought progresses, with the waterbodies progressively assuming the characteristics of the groundwater. However, during a prolonged and severe drought, the water table will continue to fall and eventually loose connectivity to the river of lake, producing a break point and sudden likely change in water quality. This can also lead to fragmentation of headwaters and stagnation of ephemeral rivers, resulting in sudden physical and chemical changes in water quality. Other sudden step changes in water quality can be triggered by lake stratification and sediment anoxia and related releases of pollutants and nutrients. The impacts on algae, cyanobacteria and the wider aquatic ecology are relatively uncertain. It is largely assumed in the scientific literature that algae and plant growth rates and biomass will tend to increase during droughts, but relationships between flow, water temperature and the interacting components of aquatic communities are difficult to predict.

To address the uncertainties in water quality and phytoplankton discussed above, the water quality of the River Thames and two of its contrasting tributaries were investigated (2009 – 2022), to determine the impacts of the 2022 summer drought. Water temperatures were the highest observed in all three sites in summer 2022. Total phosphorus (P) and dissolved organic forms of P were elevated in the Thames. The drought had little impact on nitrate and ammonium concentrations, and suspended sediment concentrations remained low. Dissolved oxygen concentrations steadily declined in the Thames through July and August 2022, dipping to 70% following a small algal bloom crash. The summer 2022 drought produced the highest observed chlorophyll and diatom concentration in the lower Thames, and a community shift towards smaller-sized pico-chlorophytes and cyanobacteria during this period of low flows and high water temperatures.

Targeted and comprehensive monitoring needs to be in place to capture the impacts of future droughts, from pre-drought conditions through to the recovery period. It would provide robust data for setting up and further-developing models. It would also provide the much-needed evidence to determine if the predictions of potential drought impacts are accurate, and in which waterbodies they occur. It would allow the characteristics of drought-resilient and drought-vulnerable catchments to be identified, which would inform catchment managers how to increase resilience in vulnerable catchments through land-use change, water quality improvements and river restoration.

1. Introduction

UK freshwaters are under increasing stress from population growth, agricultural intensification, inputs of an ever-growing range of emerging contaminants, and climate change. Current climate change projections for the UK predict wetter winters and drier, warmer summers, which are likely to increase the incidence of summer droughts, and these droughts are likely to cover a greater spatial extent, and be longer in duration (Johnson et al., 2009). Extreme low flows and long dry periods are predicted to have major impacts on water quality, as transfers of nutrients and pollutants from the catchment land surface to the waterbody will be minimised, and the impact of point source inputs will tend to be magnified due to the lack of dilution in the river or low flushing rates in lakes. The long residence times and often increased water temperatures and nutrient concentrations can result in high algal and bacterial biomass and rapid biogeochemical cycling in rivers and lakes, which have important impacts on water quality and dissolved oxygen concentrations.

In this study, we will

- Review the latest state of understanding of the impacts of drought on water quality and phytoplankton, including drought termination.
- Determine if, and to what extent, these predictions occurred across the River Thames catchment during the 2022 summer drought.
- Identify knowledge gaps that could help us to manage water quality and resources more effectively in the face of increasing drought risk.

1.1. Definitions and scope

This study will address how droughts can affect river and lake water quality. It will focus on physical, chemical and biological impacts, as these are strongly interlinked within waterbodies and across their catchments. The study will focus on recent (post 2010) drought monitoring and modelling studies from the UK and internationally, building on a previous review by Whitehead et al. (2009). It will also include low-flow events, and not be limited to official droughts. We cover parameters of environmental concern, such as macronutrients (phosphorus and nitrogen), metals, pollutants and solutes, electrical conductivity, pH, dissolved oxygen, sediments, and phytoplankton.

We also address water temperature, as hydrological drought conditions and their often-associated high air temperatures, can impact water quality via their direct effect on water temperature: the combination of shallower depth and slower water velocity tends to lead towards higher stream temperatures (Booker and Whitehead, 2022). Higher water temperatures directly impact water and sediment chemistry, dissolved oxygen concentrations, and biological biomass and metabolic activity, which can all have major impacts on water quality.

The second part of this study is a case study that reviews water quality data from the River Thames and two contrasting tributaries through the 2022 summer drought, to determine the magnitude and extent of the drought impacts that have been predicted in the scientific literature.

2. Drought impacts on river and lake water quality

Drought periods and their associated lack of precipitation result in low river flows, decreasing lake volumes and a reduced connectivity with the surrounding catchment and groundwater. This can have significant impacts on the relative sources of pollution and nutrients reaching the waterbody, and thereby affect concentrations and the chemical form of pollutant. This impact is likely to vary, depending on the hydrology, morphology, catchment land-use and pollution sources.

2.1. Nutrients

Drought and low flow

Rivers that are dominated by diffuse, rain-related inputs of nutrients (phosphorus and nitrogen) will tend to have reduced loads and/or concentrations throughout a drought period, due to the reduction or cessation of catchment inputs. Rich sources of nutrients, such as fertilisers and animal manures will not be transported from the catchment surface into rivers or lakes via field drainage, overland flow etc. Groundwater inputs (which can be a major source of nitrate in the UK) can also reduce as the water table falls, but the proportion of the total river flow that constitutes groundwater may increase as other water sources decrease. Other diffuse inputs of nutrients will be derived from subsurface flow and release from internal sources such as river/lake sediments and organic matter decomposition.

In contrast, rivers dominated by nutrient inputs from sewage treatment works (STW) effluent will tend to have increased nutrient concentrations during drought low-flow periods, because the STW point source inputs remain relatively constant, and the reduced river flow means that there is less baseline dilution capacity (Bowes et al., 2014; Naden et al., 2015). The relationship between nutrient concentration and river flow for the Thames catchment, and how this varies according to the proportion of STW effluent inputs, is shown in Figure 1. These weekly data from 2009 to 2014 did not include an official drought period, but show how tributaries receiving significant STW point source inputs have higher total phosphorus (TP) concentrations as flows reduce, whereas catchments that have low population densities and few sewage inputs (River Pang) have declining TP concentrations as flows decline.

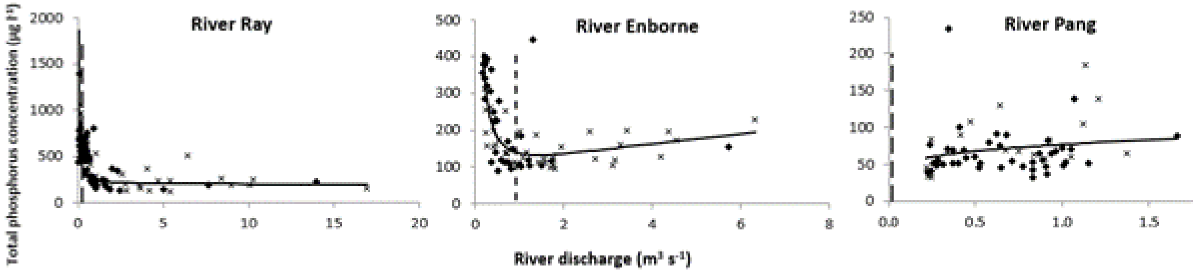

Figure 1. Typical total phosphorus concentration / flow relationships derived from STW point source dominated (R. Ray, Oxfordshire), mixed source (R. Enborne) and diffuse dominated (R. Pang) catchments (Naden et al., 2015).

Nutrient concentration is not merely a product of the load of nutrient being input into a waterbody and the dilution effect of the river flow or lake flushing rate. Nutrient dynamics within rivers and lakes are complex and interacting, including processes such as biological uptake, sequestration by sediments, and transformation into different nutrient forms by chemical and biological reactions. These processes tend to be at their maximum during low flow periods, when water residence time is at its maximum, nutrient concentrations can be at their peak and water temperatures often high. Under these extreme conditions, macrophytes, periphyton and phytoplankton can grow rapidly, resulting in the depletion of phosphorus, nitrogen and dissolved silicon from the water column and bed sediment. The low flow conditions, with reduced water depth and slower flow velocities, results in dissolved phosphorus being in greater contact with the bed, thereby potentially increasing the rate of adsorption to the sediment and loss from the water column (Bowes and House, 2001). Consumption of aquatic biomass / organic matter by microbes, and algal respiration, can reduce oxygen concentrations, particularly at night, and cause anoxic conditions in bed sediments and the lower water layers. This can result in sudden releases of phosphorus and ammonium into the overlying water, thereby increasing the risk of eutrophication (Jarvie et al., 2020). These effects are particularly marked in thermally-stratified lakes, where the lower layers of the lake become anoxic.

Drought periods and high temperatures also affect the uptake and release of nutrients from the catchment soils. High temperatures can increase organic phosphorus soil mineralisation, and soil drying can result in phosphorus release via the oxidation of soil carbon, aluminium and iron (Forber et al., 2018). Conversely, other studies have shown that mobile nutrients are more likely to be retained within soils during drought periods (Costa et al., 2022).

Drought cessation

Catchment characteristics and relative nutrient sources can have a large impact on water quality during the first storm events following a low flow or drought period (Outram et al., 2016). The previous prolonged period without rainfall means that nutrients from animal manures, fertilisers and organic matter will have accumulated on the catchment surface. The first rainfall events often rapidly-mobilise this stored nutrient and transport it into the waterbody, thereby increasing the loads and potentially the concentrations. Long-term water quality monitoring studies, such as the Defra Demonstration Test Catchments and UKCEH Thames Initiative have shown large peaks in phosphorus concentration during these first storms following a prolonged dry spell (Bowes et al., 2015a; Bowes et al., 2015b; Ockenden et al., 2016), and similar patterns have been observed elsewhere for phosphorus (Bieroza et al., 2019; Lisboa et al., 2020) and nitrogen (Loecke et al., 2017).

In river catchments receiving STW final effluent discharges throughout the low flow or drought period, the first storms often produce large clockwise hysteresis loops, with much higher than expected phosphorus concentrations on the rising hydrograph for a given river flow (see purple section of Figure 2, lower graph). These data points do not coincide with increased inputs of nitrate or ammonium, and they are therefore from a rapidly-mobilised source, in close proximity to the river monitoring point, that consists primarily of phosphorus and not nitrogen. This suggests that much of the continuous phosphorus inputs from STW effluents over the low-flow period are stored within the bed-sediment, and the first few storm events disturb the bed, mobilising phosphorus-rich sediments and pore-waters into the water column. The subsequent storm events often result in lower P concentrations as this source of rapidly-mobilised bed-sediment P becomes depleted and catchment P inputs become more significant as the catchment connectivity with the river increases as it wets-up (brown section in Figure 2; lower graph).

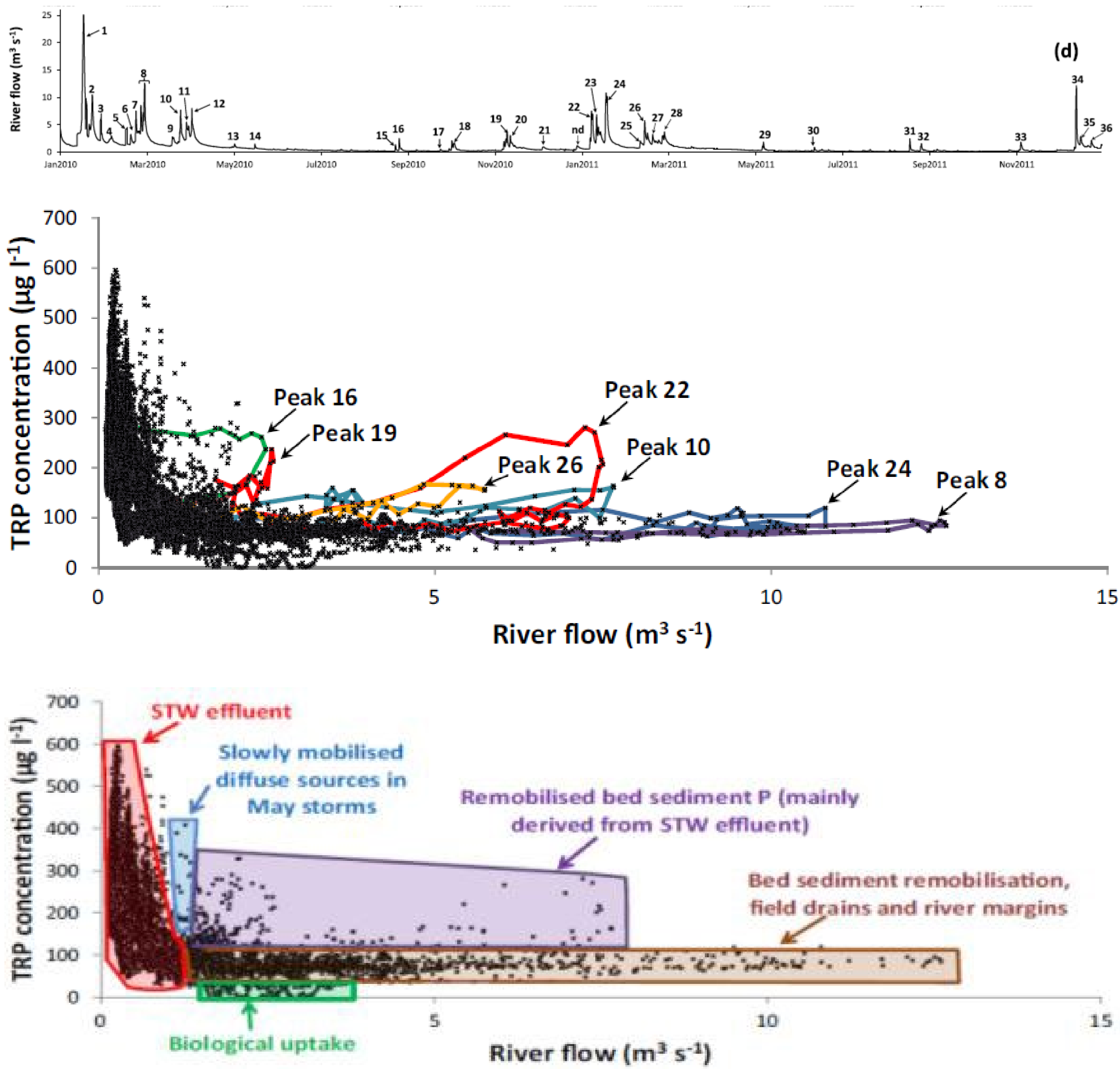

Figure 2. Total reactive phosphorus concentration / flow relationship for the River Enborne, Berkshire (Bowes et al., 2015b)((Bowes et al., 2015b).

Nutrient modelling

The relationship between river nutrient concentration and flow can be modelled, using the Load Apportionment Model (LAM) (Bowes et al., 2008), to estimate the relative quantities of P and N coming from continuous (point source) and rain-related (diffuse) sources. These established concentration/flow relationships can then be used to estimate the nutrient concentrations resulting from low flow and drought conditions. This LAM method was applied to the Future Flows Climate dataset, which provided daily mean flows for 115 rivers across England, for 11 climate predictions generated by the Hadley Centre Regional Climate Model (HadRM3-PPE) (Prudhomme et al., 2012). This enabled phosphorus concentrations to be predicted under these 11 climate scenarios, at daily resolution, until 2098 (Charlton et al., 2018). This Environment Agency study predicted that for some rivers, there would generally be small increases in annual ortho-phosphorus concentrations, and increases in summer concentrations in particular, due to climate change and associated reduced summer flows, and this could increase future eutrophication risk (Environment Agency, 2016). A modelling study of the Seine River in Paris predicted average increases of 19% and 32% in nitrate and phosphate respectively during extreme low flow periods, (Raimonet et al., 2018).

Various INCA (Whitehead et al., 1998) modelling studies have also suggested that climate change projections for the Thames catchment are likely to have very little impact on predicted nitrogen (Jin et al., 2012) and phosphorus (Wade et al., 2022) concentrations and loads. Others Thames studies have suggested that by 2050, nitrate may reduce due to increased in-stream denitrification rates and phosphorus concentration would increase by 12% due to reduced dilution of effluents (Bussi et al., 2017). The INCA-N model has also been applied to the rural River Wye in Wales, which predicted little change in the upper catchment, but increased nitrate concentrations and loads in the lower catchment (Bussi et al., 2018).

2.2. Metals

There are many potential sources of metal inputs to rivers and lakes, including the natural weathering of bedrock and soils, anthropogenic sources from road and agricultural runoff, and point source inputs from sewage and industrial effluents (Pulley et al., 2016). Metal contamination of rivers can be potentially toxic to aquatic biota and result in failure to meet regulatory water quality targets, particularly in the many areas of high population densities and historic mining activity across the UK.

Droughts and low flows

The impact of low flows on heavy metal concentrations and seasonality will vary greatly, depending on catchment sources and which metallic element is being observed. Where the major source of metal pollution is from continuous sewage and industrial effluents, waterbodies will have highest metal concentrations during low flows, when dilution is at a minimum. Areas with abandoned metal mines may also have significant metal inputs during low flows, due to the relatively high proportion of metal-rich groundwater and soil leaching (Byrne et al., 2020). However, under severe drought conditions, these groundwater inputs may cease as water tables drop, reducing the river concentration and load. Conversely, periods of rainfall will wash metals from land and road surfaces, and mobilise mine waste from soils and spoil heaps, thereby increasing river loads as flow increases.

Dissolved metal sorption by waterbody bed-sediments converts dissolved metals to particulate-bound forms. Contact between dissolved metals and bed sediments will be highest under low flow conditions, due to low flow velocities, high residence times and a greater relative contact with the bed as river levels fall. However, this flow-related pattern is further complicated by redox and pH effects, with rises in pH during droughts (e.g. due to reduced inputs from upland acidic soils and peat) resulting in possible release of metals from the sediment to the overlying water column (Byrne et al., 2020; Pulley et al., 2016). This complexity in metal source, speciation and internal geochemical cycling means that there is no clear pattern in metal concentration during drought conditions. However, a literature review by Mosley (2015) showed that droughts usually resulted in increased metal concentrations in both rivers and lakes. A study of the River Meuse in Belgium showed that during drought conditions, metals with high adsorption affinity with sediments (such as lead, chromium, mercury and cadmium) had low river water concentrations, whereas those metals with low adsorption affinities (selenium, nickel, barium) had increased concentrations at low flows (van Vliet and Zwolsman, 2008). The reduced lake flushing rates and higher water temperatures associated with droughts can increase incidence and duration of lake thermal stratification. This can lead to anoxia in the hypolimnon, which has been shown to release metals such as iron and manganese from the bed sediment into the overlying waters (Mosley, 2015).

Drought cessation

The storage of metals within the bed sediment of rivers and lakes during low flow and drought conditions, through absorption of dissolved metals and sedimentation of particulate-bound metals can provide a major source for metal remobilisation when the drought comes to an end. Large increases in metal concentrations in the first storms following a drought have been observed (Hrdinka et al., 2012), but there have been very few studies investigating drought recovery of stream biogeochemistry (Lake, 2011).

2.3. Solutes and pollutants

Rivers and lakes receive a complex mixture of substances from domestic and industrial wastewaters, and diffuse inputs from groundwater, soils, agriculture, septic tanks and runoff from urban surfaces. This vast mixture of solutes and pollutants, derived from a wide range of sources with a variety of transport pathways and chemistries, will all react differently to droughts. They will also vary spatially, based on waterbody type, catchment characteristics, physical and chemical conditions, biological interactions and hydrological connectivity.

It would be impossible to predict drought impacts for such complex systems, but generalisations can be made. Pollutants that are primarily routed into rivers and lakes via sewage treatment works final effluent, such as pharmaceuticals, personal care products, synthetic hormones, plasticisers, flame retardants and microplastics, are likely to be at their highest concentrations during low flows (Kamjunke et al., 2022; Watkins et al., 2019), and these concentrations are predicted to increase with increasing drought frequency and intensity in the future (Whelan et al., 2022). In contrast, pollutants that are associated with diffuse pollution, such as pesticides, herbicides and road runoff, would be expected to be at low concentrations during drought periods, due to lack of connectivity between the catchment surface and the waterbody. At drought cessation, it would be expected to see these accumulated diffuse pollutants producing significant concentration peaks within the waterbody.

Electrical conductivity (EC) provides a proxy for inorganic solute concentrations within a waterbody, and can be a good indicator of pollution concentration. Studies have shown that during severe low flows, EC is elevated (Burt et al., 2015; Hellwig et al., 2017; Jones and van Vliet, 2018), due to lack of dilution of sewage effluents, and increasing solute concentration through evaporative losses. Other studies have also observed reduced pH and acidification during droughts (Jones and van Vliet, 2018; Mosley et al., 2014), but many monitoring studies have shown no pH impact (reviewed in Mosley (2015)).

2.4. Dissolved oxygen

Droughts in the UK are often associated with periods of warm weather, and the combined effect of low flow and increased water temperature can result in low dissolved oxygen (DO) concentrations in waterbodies (Whitehead et al., 2009). Low DO can cause problems for the fish (Warren et al., 2015) and invertebrate (Parr and Mason, 2003) communities, and the potentially anoxic conditions can also result in poor water quality (Gomez-Gener et al., 2020).

The solubility of oxygen in water decreases with increasing temperature, and therefore DO concentration would be expected to decrease as water temperature rises. Low flows result in reduced turbulence, which limits re-aeration of the waterbody from the atmosphere (Warren et al., 2015). In severe droughts, longitudinal connectivity of rivers can be lost, particularly in ephemeral headwater streams, and this can cause significant reductions in DO in these stagnant waterbodies (Dollar et al., 2012). Increasing temperatures can also lead to increases in bacterial concentrations and rates of respiration. Additionally, the low flows and lack of dilution of sewage-derived organic pollution sources and high biological productivity within the waterbody can result in high biological oxygen demand (BOD), which can further reduce DO concentrations (Johnson et al., 2009). Algal and plant photosynthesis can be greatly elevated during summer drought periods, due to increased residence time resulting from the reduced flows, and the associated sunny and warm conditions that often occur, and this can counteract any reductions in DO. However, when the algal bloom ends, respiration dominates (due to the sharp reduction in photosynthesis and the bacterial consumption of algal biomass), and this can result in oxygen sags (Martin et al., 2013). In lakes, increasing water temperatures and reduced flushing rates can increase lake stratification, which can often lead to oxygen depletion in lake hypolimnia (Hering et al., 2010). The review by Mosley (2015) showed that droughts caused reductions in DO in rivers and lakes in most cases.

Routine regulatory monitoring of rivers and lakes during the day usually records oxygen oversaturation (DO% >100%) during these periods. However, high-frequency sub-daily monitoring shows that there are large diurnal fluctuations in DO concentration during periods of high algal biomass, which could lead to low DO and associated ecological problems overnight (Halliday et al., 2015), such as loss of invertebrate taxa and increased fish mortality (Chessman, 2015; Warren et al., 2015). Low night-time oxygen concentrations are particularly apparent in headwater catchments, for example in urbanising areas (McGrane et al., 2017) where problematic levels can be seen in colder conditions as well as in summer. In small rural rivers similarly low concentrations are observed (e.g. Hutchins et al. (2021)). Present day summer DO concentrations in larger rivers are higher (typically 8.0-8.5 mg/L outside of depleted conditions during bloom crashes). A review of response to droughts showed mixed effects to dissolved oxygen (Mosley, 2015). A few studies revealed DO increases attributable to enhanced primary productivity, whereas for the majority, declines are reported due to increased water temperature and these are especially apparent in point source dominated systems. Following droughts, the delivery of accumulated labile carbon in post-drought flushes can give rise to further deoxygenation.

Dissolved oxygen modelling

Process-based modelling studies can quantify impacts of droughts on oxygen levels and offer mechanistic explanations. In addition to simulating seasonal variations, the QUESTOR model reproduces observed diurnal fluctuations of dissolved oxygen and key drivers (water temperature and chlorophyll) successfully (Hutchins et al., 2021; Pathak et al., 2021). Elevated photosynthesis during algal blooms enhances dissolved oxygen but depletion can occur following crashes in prolonged dry periods (Waylett et al., 2013). In a model sensitivity analysis of river water in the south-east, Hutchins and Hitt (2019) addressed the effects of reductions in flow and increases in water temperature which are key features of UK summer droughts. It was found that reduction in river flow by 20% in summer will likely deplete dissolved oxygen by up to 0.9 mg l-1. Separately, an increase in summer water temperature of 2 ºC will deplete dissolved oxygen by up to 0.5 mg l-1. Modelling suggests the combined effects are less than the additive expectation but would still deplete levels by more than 1.0 mg l-1.

Prediction of water quality under projected future climate conditions (derived from HadRM3-PPE data) (Prudhomme et al., 2012) reveals a likely worsening of dissolved oxygen conditions, with increased incidence above present day frequency of concentrations below 6 mg l-1 by 7-13 days per year in the River Thames. Such future deteriorations appear more likely in drier parts of the south-east where increased incidence of undesirable conditions broadly equates to reduction in typical summer DO levels by as much as 3 mg/L. In other parts of the country (e.g. rivers in Yorkshire and south-west) change in summer DO is likely to be less severe (< 1 mg l-1) (Miller and Hutchins, 2017). In summary, much of the adverse effects of droughts and likely future changes (attributable to the integrated effects of drier and warmer conditions) are manifested in terms of consequences of algal bloom crashes, enhanced heterotrophic respiration and reduced oxygen holding capacity of warmer waters.

Raimonet et al. (2018) predicted the oxygen sags would occur and increase in magnitude at the end of spring algal blooms in the lower River Seine during extreme low flows. Here, models also identified the predominance of benthic processes in controlling carbon dynamics and enhanced oxygen consumption at low flows (Vilmin et al., 2016). Deep learning models also provide insights and can successively predict low levels in rivers (< 7.5 mg l-1) under high temperatures typical of droughts and where oxygen consuming in-channel transformations are identified (Zhi et al., 2021). However oxygen response in rivers is closely related to river hydrodynamics and where such information is lacking this can hamper prediction of low concentrations in droughts.

2.5. Sediment

Suspended sediment concentrations in waterbodies are closely related to rainfall, river flow velocities and lake flushing rates. Rainfall and snowmelt allow surface sediment to be washed into the river or lake. The higher river flows, lake flushing rates and turbulence keep more of the sediment inputs suspended in the water column. The higher flow velocities re-suspend finer bed-sediments. Therefore, during droughts, sediment inputs to the waterbody and bed-sediment resuspension will be expected to be at a minimum (Hrdinka et al., 2012). Low flow velocities will increase the rate of deposition, and result in finer, organic-rich sediments being deposited on the bed (Dollar et al., 2012). This can have major impacts on dissolved oxygen and water quality, as the increased microbial biodegradation of this fine organic matter, under increased water temperature conditions of the drought, can cause anoxia of the bed sediment (Mosley, 2015), leading to phosphorus and ammonium releases to the overlying water (Jarvie et al., 2020; van Vliet and Zwolsman, 2008).

The first storms following a drought period can produce high suspended sediment loads (Bieroza et al., 2019; Burt et al., 2015), due to the remobilisation of fine bed sediments that have accumulated over the drought period, and due to the in-wash of material that had accumulated across the catchment.

2.6. Phytoplankton

Drought periods are generally predicted to increase algal bloom risk in rivers and lakes (Whitehead et al., 2009). Low river flows and reduced lake flushing rates produce longer residence times that allow phytoplankton biomass to increase (Bowes et al., 2012; Bowes et al., 2016; Forber et al., 2018; Reynolds, 2006). The warmer temperatures and increased light availability that are often associated with droughts are also predicted to increase gross primary production in rivers and lakes (Desortova and Puncochar, 2011; Hosen et al., 2019). Increased phytoplankton biomass can increase water column turbidity and excessive epiphytic algal growth can cover plant leaves, resulting in macrophyte community shifts towards emergent species, or the potential loss of macrophytes, due to light limitation (Hilton et al., 2006; O'Hare et al., 2018). This shift or loss of plant habitat can have damaging impacts on the invertebrate and fish communities.

Conversely, very low flow-velocities and low turbulence can result in the sedimentation of the larger phytoplankton and a reduction in biomass within the water column (Kamjunke et al., 2022). In a 30-year study of the Loire River, France, Minaudo et al. (2021) showed that despite increases in water temperatures and extreme low-flow periods, phytoplankton biomass has decreased 10-fold, which was attributed to major reductions in phosphorus concentrations and increased grazing rates by the presence of an invasive filter-feeding clam. High water temperatures in the River Thames have also been shown to regularly end chlorophyll blooms (Bowes et al., 2016). However, the majority of recent studies have generally observed increased algal biomass in both rivers and lakes as a consequence of drought (Mosley, 2015).

The community composition of phytoplankton is also likely to be affected by climate change. The low flow conditions during droughts can shift the phytoplankton community towards smaller chlorophytes and cyanobacteria, as the larger phytoplankton, such as diatoms, settle out of the water column. Cyanobacteria are known to often dominate the phytoplankton community when water temperatures are high (Paerl and Huisman, 2008), and are also relatively salt tolerant, and so not affected by increasing conductivity due to evaporation loses (Reichwaldt and Ghadouani, 2012). Most recent studies have observed increases in cyanobacterial biomass in lakes (Havens et al., 2019; Ji et al., 2017) and rivers (Piano et al., 2017) during drought periods, but some studies have not seen any impact (Minaudo et al., 2021).

Modelling

A 5-year niche-modelling study of the middle River Thames, southern England, has shown that chlorophyll blooms occur during periods of low flow, high sunlight and warm water temperatures (Bowes et al., 2016). However, chlorophyll blooms collapse when flow is exceptionally low (<10 m3 s-1) and water temperatures exceed 19oC. Therefore, developing droughts could significantly increase phytoplankton biomass, but droughts with extremely low river flows and potentially high temperatures could actually decrease phytoplankton biomass and alter the timing of blooms.

Bussi et al. (2016) applied the INCA model (Whitehead et al., 1998) to the same River Thames dataset, alongside cell abundances of five phytoplankton groups, which predicted that average phytoplankton concentrations were likely to increase under future climate change scenarios, with cyanobacteria in particular showing a significant increase. Raimonet et al. (2018) predicted that phytoplankton biomass will increase by an average of 31% in the lower Seine River as a result of extreme low flows and associated phosphorus and nitrate increases.

UK lake modelling studies have also predicted high abundance of cyanobacteria during drought periods with high water temperatures and low flushing rates (Carvalho et al., 2011; Elliott, 2010).

3. River Thames case study: summer 2022 drought

The weekly water quality and phytoplankton data from the UKCEH Thames Initiative (Bowes et al., 2018) was used to assess the impact of the 2022 drought in southern England. This provides an opportunity to determine if the patterns and predictions made in the recent research literature was accurate for this particular catchment, and how the impact varied in rivers of different scales, population density, and land use.

This case study focussed on the lower River Thames at Runnymede, and two contrasting tributaries; The Cut and the River Kennet (Figure 3; Table 1). The Thames at Runnymede has a relatively high population density (371 people km-2), and flows through the major urban areas of Swindon, Oxford, Reading and Maidenhead. It is therefore nutrient enriched from the large amounts of STW effluent it receives. Much of the catchment is rural, dominated by arable land use. The Cut is a highly urbanised (1644 people km-2), highly modified river with low base flow index (BFI). The River Kennet is a largely arable, groundwater-fed chalk stream, with high base flow index and lower population density (114 people km-2). In addition, we also utilise the water quality data from the EA / UKCEH automatic monitoring station on the River Thames at Taplow, Maidenhead (Figure 3). This provides hourly data for water temperature, conductivity, pH, ammonium, turbidity, dissolved oxygen, chlorophyll, and nitrate.

Table 1. Catchment statistics for Thames study sites.

River name / monitoring location		River Kennet at Woolhampton	The Cut at Paley Street	River Thames at Runnymede
Grid reference		SU572667	SU869762	TQ006723
Catchment area (km^2)		845	63	7170
Distance to source (km)		79	21	240
Base flow index		0.87	0.46	0.72
% Land cover	Grassland	30.0	32.7	34.0
	Arable	48.7	9.8	40.4
	Urban / semi-urban	4.4	35.3	10.5
STW PE		96,380	103,600	2,661,370
STW PE density ($PEkm^{-2}$)		114	1644	371

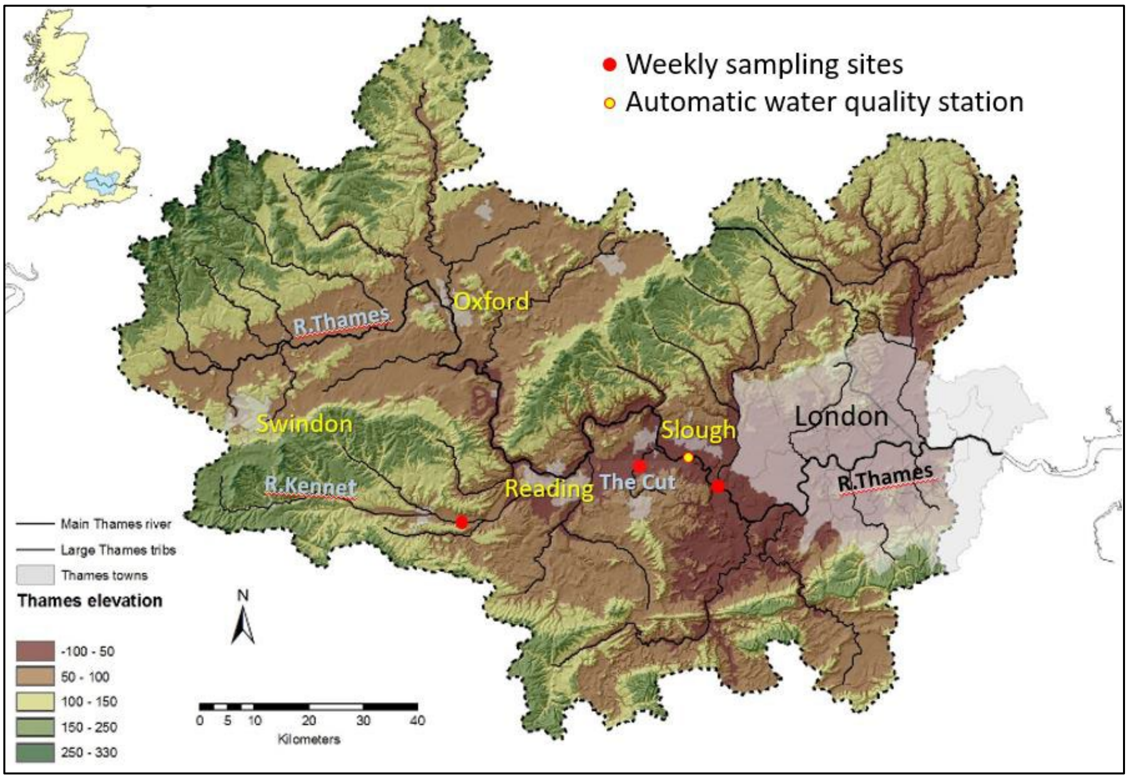

Figure 3. Map of River Thames catchment with location of weekly monitoring sites on The Cut at Paley Street, Kennet at Woolhampton and Thames at Runnymede (red circles) and automated water quality monitoring station at Taplow (yellow circle)

3.1. Water temperature

All three rivers had their highest recorded water temperatures during the 2022 drought, all peaking on the same sampling day; 18th July (Figure 4). Maximum observed temperatures varied greatly between the study rivers, with the highest being in the lower Thames (26.9oC), followed by The Cut (23.2oC) and River Kennet (21.3oC). The lower temperatures in the Kennet are probably related to the greater input of cooler groundwater to this Chalk river.

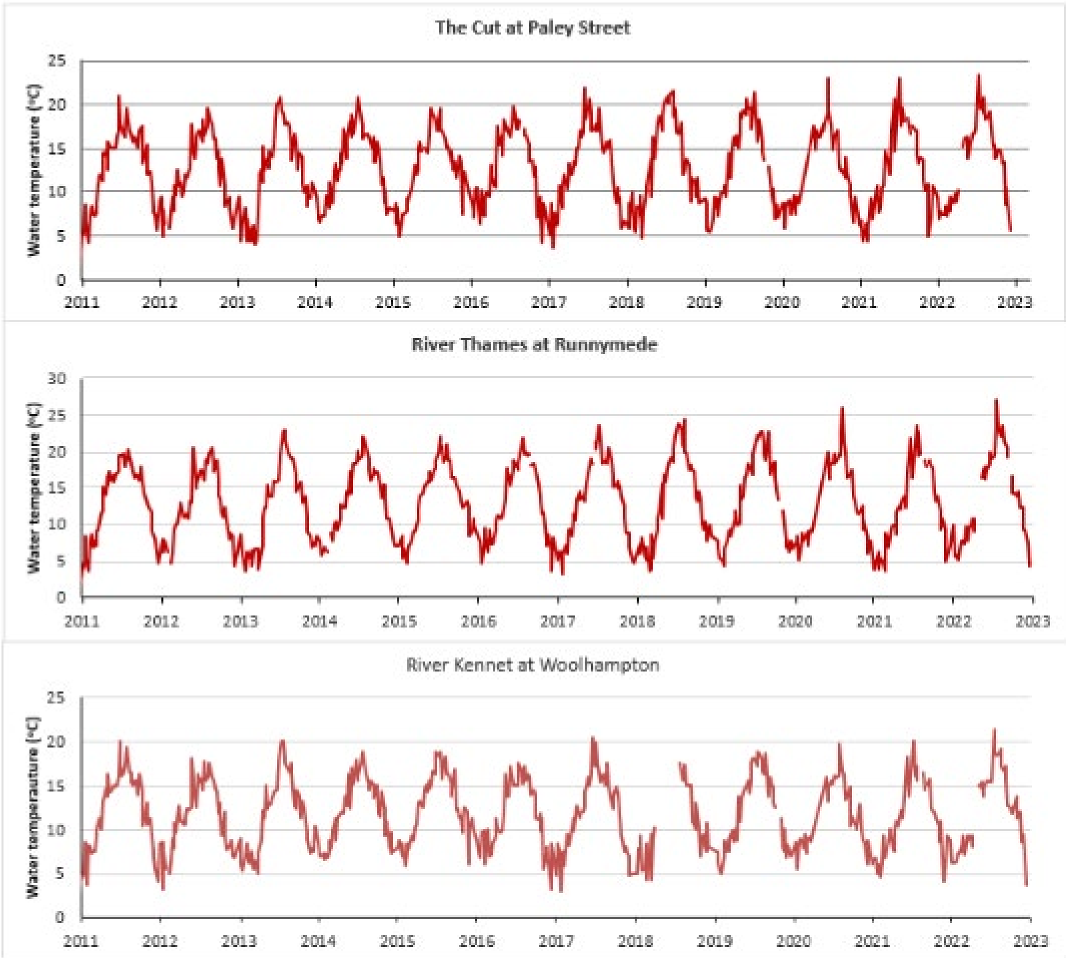

Figure 4. Weekly river water temperatures at Thames case study locations.

Therefore, the predictions that drought periods (especially if they tend to coincide with periods of hot weather) will increase river water temperature (Booker and Whitehead, 2022) appears to be correct in the Thames catchment. The pattern in this observational data suggests that drought temperature impacts will increase as river length increases and BFI decreases.

3.2. Nutrients

Phosphorus

Most of the current literature and modelling studies predict that phosphorus concentrations, particularly in urbanised catchments, would increase during drought periods, due to a lack of dilution of continuous sewage effluent inputs. The data from the study rivers confirm that both The Cut and the River Thames do indeed have a regular annual pattern with highest P concentrations in the summer and early autumn of each year (Figure 5). The only exception to this is in 2011 – 2012, where the pattern is reversed due to the winter 2011 drought and 2012 summer floods. However, the drought in summer 2022 did not produce the magnitude of peaks that would be expected during such a low-flow period. Soluble reactive phosphorus (SRP) concentrations were relatively typical of previous summer concentrations, although total P concentrations in the Thames were much higher than previous years. This suggests that the phosphorus load, and the dissolved P load in particular, is being reduced by within-channel processes, such as uptake by bed-sediments, bioaccumulation by macrophytes and algae, and deposition of colloidal phosphorus. As expected, the River Kennet did not produce a seasonal pattern in phosphorus, as its P inputs are predominantly from diffuse sources, rather than point source sewage effluents.

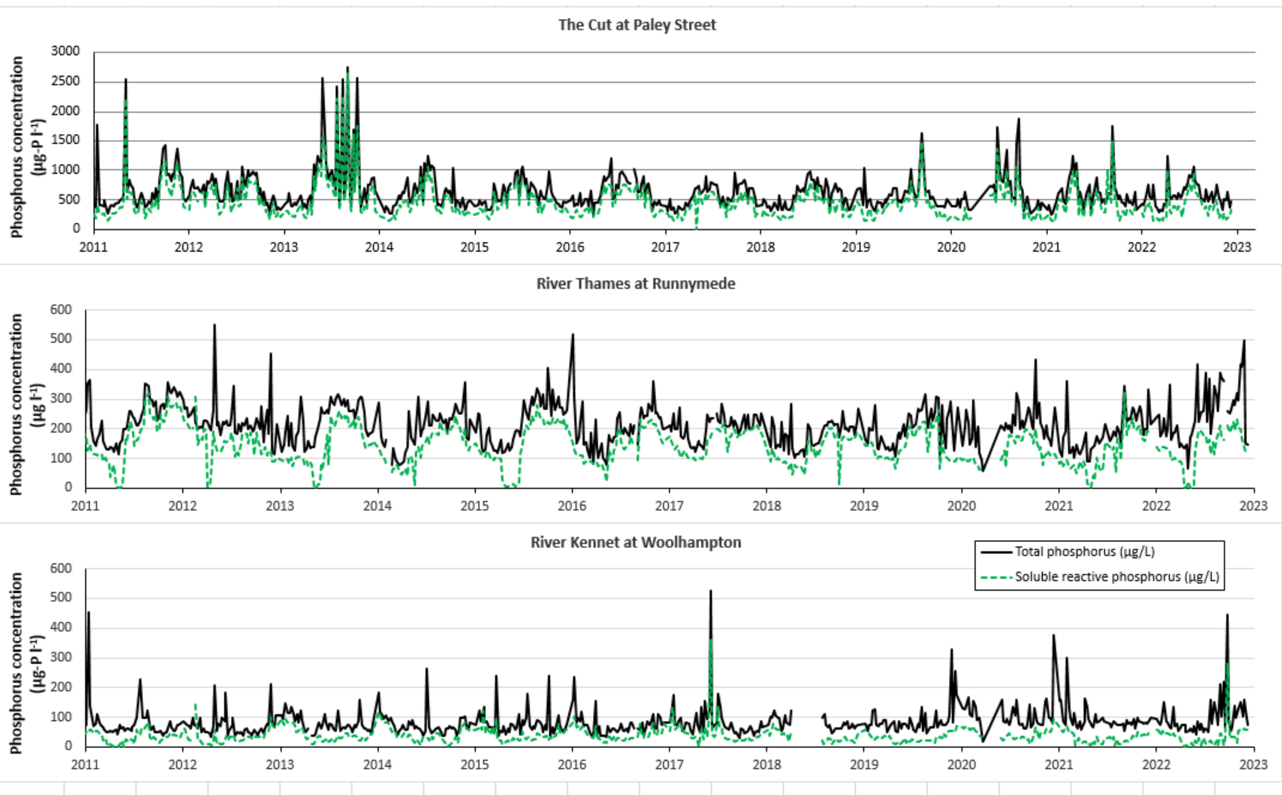

Figure 5. Weekly total phosphorus and soluble reactive phosphorus concentrations at Thames case study locations.

At the drought cessation, which occurred with some showers in September 2022, followed by heavier rain and some flooding into December 2022, there were major TP peaks of 497 and 445 µg l-1 in the Thames and Kennet respectively. This probably resulted from the transport of P that had accumulated across the catchment during the long period of dry conditions to the river, and the mobilisation of P that had been stored within the channel itself. This supports the conclusions of past UK low-flow studies (Bowes et al., 2015a; Ockenden et al., 2016). The TP concentration in The Cut did not peak during this drought cessation period. This is likely to be due to The Cut having a small catchment, which would minimise diffuse P inputs, a short distance to source, meaning limited area for storing in-channel phosphorus, and its P load continuing to be dominated by point source STW inputs. The Cut is also heavily modified and canalised in sections, which could reduce its connectivity with the floodplain and limit the areas for sediment accumulation.

The form of phosphorus changed during drought conditions. The concentrations of dissolved hydrolysable phosphorus (DHP, equivalent to dissolved organic phosphorus; calculated by subtracting the SRP from the total dissolved P (TDP)) was elevated during the 2022 summer drought for both the River Thames and The Cut (Figure 6). This suggests that available nutrients are being rapidly taken up by biota, and re-released in organic form due to elevated rates of microbial decomposition during the high-temperature and low-flow conditions.

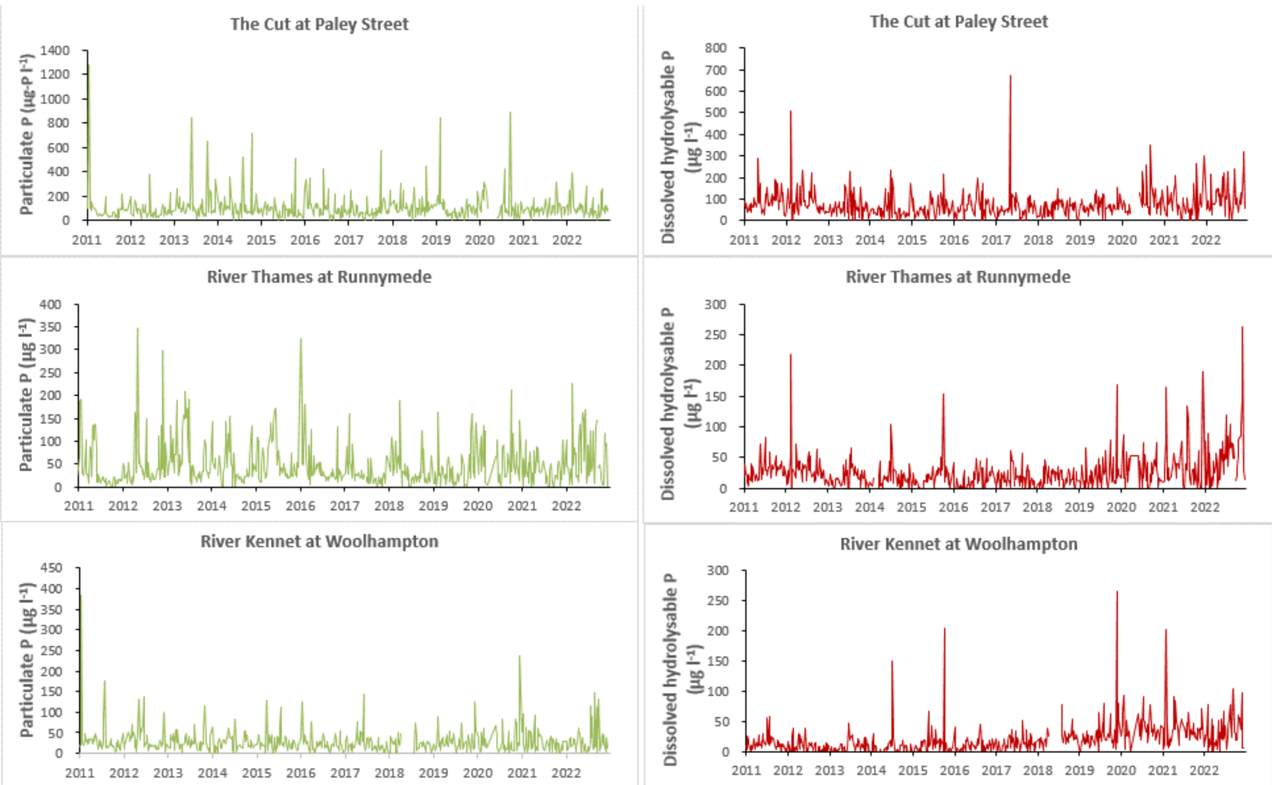

Figure 6. Weekly particulate-bound phosphorus and dissolved hydrolysable phosphorus (DHP) concentrations at Thames case study locations.

The River Thames had high concentrations of particulate phosphorus (PP; calculated by subtracting TDP from TP) of up to 170 µg-P l-1 during the 2022 drought period (Figure 6). Much of this is likely to be bio-accumulated P within the phytoplankton. During the drought cessation from September to December 2022, the River Thames and River Kennet had peaks in particulate P, probably due to the remobilisation of P-rich sediments that had been deposited onto the river bed during the period of exceptionally-low flows, and the flushing of accumulated organic material from the river channel, and input of PP from the catchment.

Nitrogen

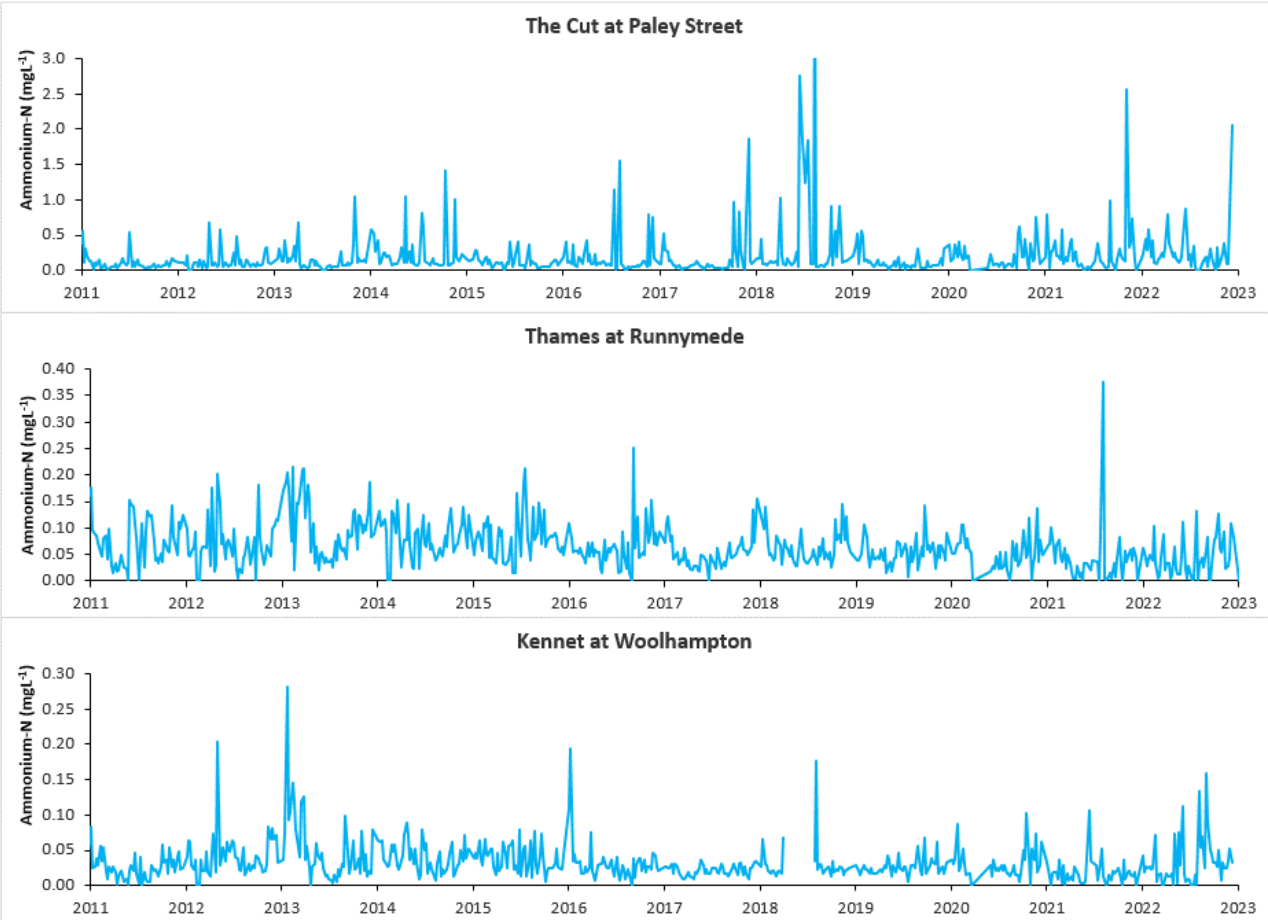

Figure 7. Weekly ammonium concentrations at Thames case study locations.

Despite predictions that ammonium concentrations can be elevated during droughts, due to the lack of dilution and release from anoxic sediments, the ammonium concentrations were extremely low at the height if the drought in August 2022 in the Thames and The Cut (Figure 8). The Kennet had very low ammonium concentrations in July, but had a peak of 0.13 mg-N l-1 in early August, which could indicate that ammonium was being periodically released from bed sediments. At drought cessation, the first rainfall in the first week of September caused a large ammonium peak (0.16 mg-N l-1) in the Kennet. In the Thames and the Cut, ammonium concentrations increased to a more typical concentration, but there were no major peaks until 12th December in The Cut, reaching 2.06 mg-N l-1.

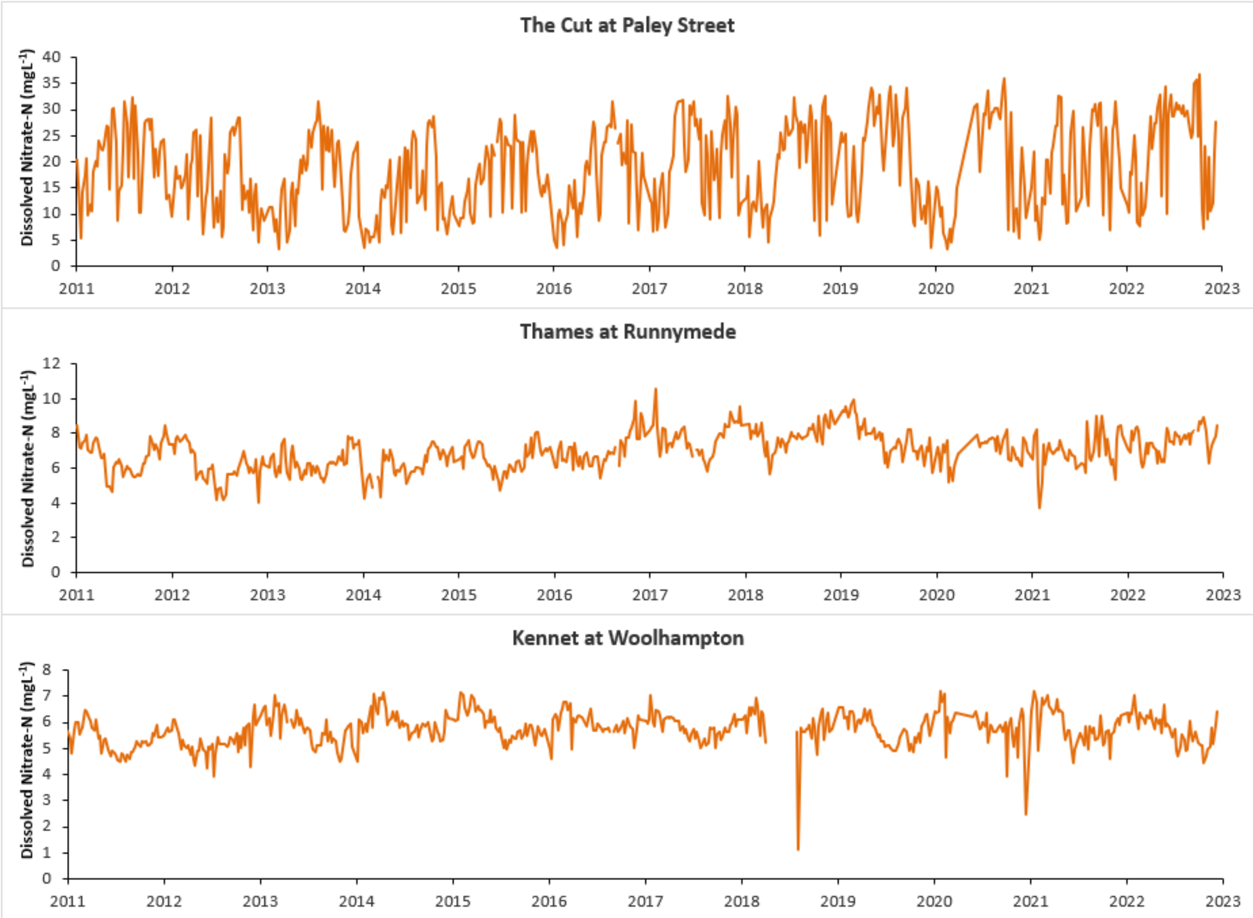

Figure 8. Weekly ammonium concentrations at Thames case study locations.

For nitrate, the time-series exhibited the annual patterns suggested in the literature, with the highest concentrations in the summer low flow periods for the sewage-dominated Thames and Cut. In contrast, the Kennet showed the opposite pattern, with highest nitrate concentrations in the winter and spring high-flow periods, due to the nitrate inputs being dominated by diffuse, agricultural sources. During the July and August 2022 period, the Thames and Cut nitrate concentrations were very steady and consistently high at approximately 8 and 30 mg NO3-N l-1 respectively). At Drought cessation in September and October, The Cut had the two highest nitrate concentrations of the entire 12-year dataset at 37 mg nitrate-N l-1, followed by a sudden reduction to 7.2 mg-N l-1 within the two weeks to the end of October 2022. This indicates that the rainfall mobilised catchment and within-river N sources, but these were rapidly exhausted, and then the extra flow was merely diluting the sewage nitrate inputs. The Thames at Runnymede showed a similar pattern with an increase in nitrate from 8 to 9 mg N l-1, and then a sudden reduction to 6.25 mg-N l-1 by the end of October. The diffuse-dominated Kennet showed the opposite pattern, with a steady increase in nitrate from the end of October as the catchment wetted up.

3.3. Solutes

Electrical conductivity showed a marked seasonal pattern in The Cut, with highest EC during summer and early autumn low-flows (Figure 9), due to the dominance of continuous solute inputs from sewage effluents. The River Thames showed a similar but less marked seasonal pattern, due to its lower population density, whereas the largely-rural River Kennet had a relatively consistent EC value. The Cut and River Thames had the highest observed EC values during the 2022 drought period, whereas EC in the River Kennet was unaffected.

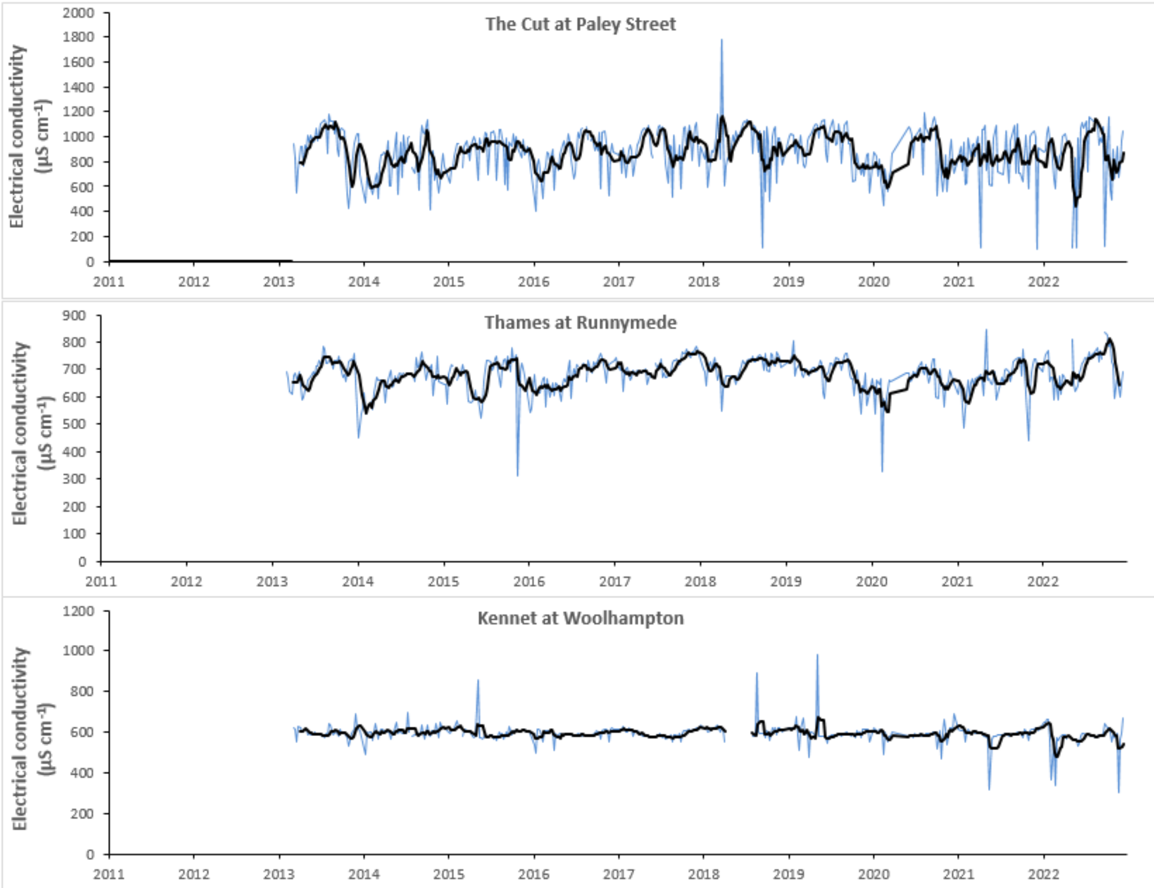

Figure 9. Weekly electrical conductivity data at Thames case study locations

The hourly EC data for the River Thames at Taplow showed an increase over the course of the drought period, from 650 to 750 µS cm-1, but EC declined during periods of high phytoplankton biomass and chlorophyll concentration (Figure 10). At drought cessation, the storm events diluted the solutes at all four study sites (Figure 9; Figure 10).

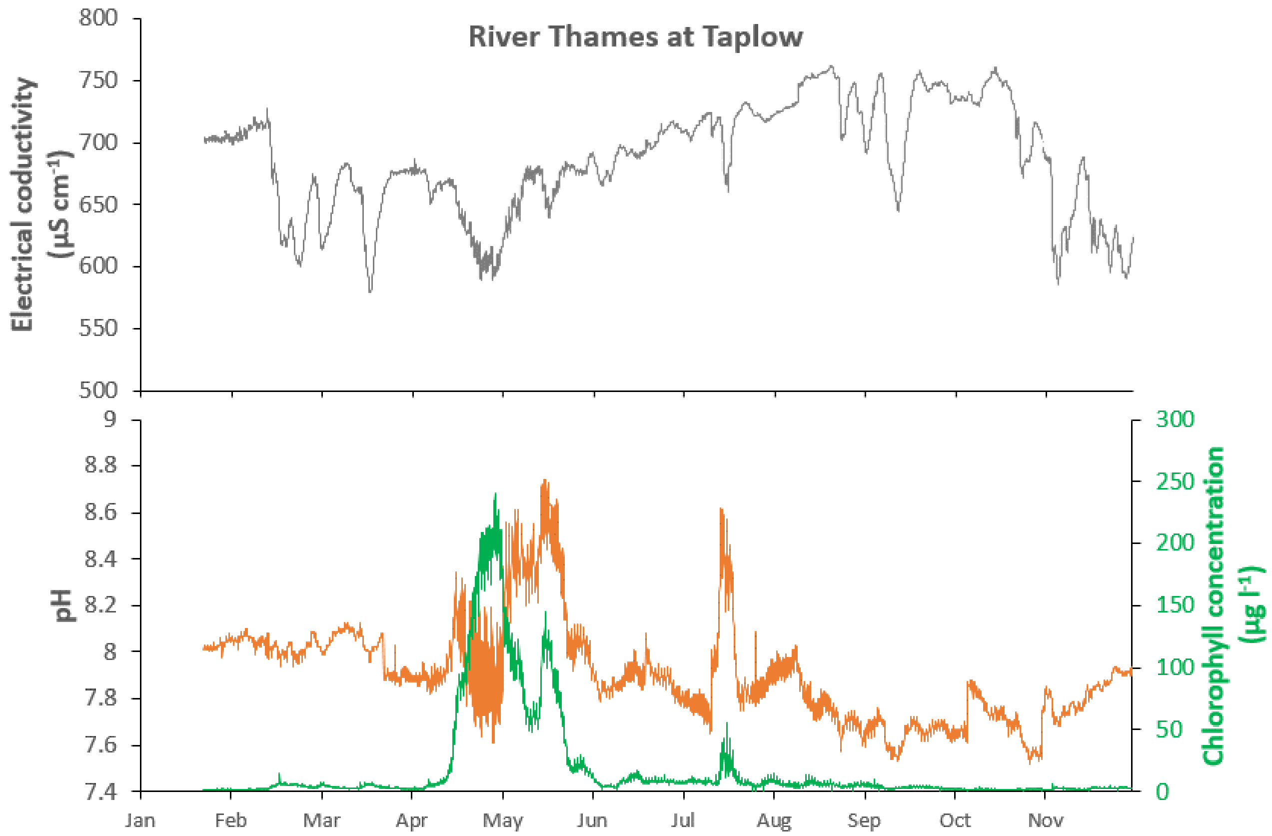

Figure 10. Hourly electrical conductivity and pH data from the River Thames at Taplow, Maidenhead

The hourly pH data of the River Thames at Taplow generally reduced during the drought period, from 7.94 at the start of April to 7.65 in late August (Figure 10), showing that the drought was increasing acidity, as observed in some previous studies. However, the two phytoplankton blooms in mid-April-late May and mid-July resulted in elevated pH values of up to 8.7, due to the high rates of photosynthesis reducing the CO2 concentration in the river.

3.4. Suspended sediment

The Cut and River Kennet study sites had very low suspended solids concentrations throughout the drought period, and the whole of 2022 (Figure 11). The Thames had a suspended solids peak of ca. 30 mg l-1 in May 2022, which corresponded with a peak in chlorophyll, indicating that the suspended solids were predominantly comprised of phytoplankton (mainly diatoms) rather than sediment. These observations support the prediction that suspended sediment load will be low through droughts, due to reduced catchment inputs and siltation within the river channel.

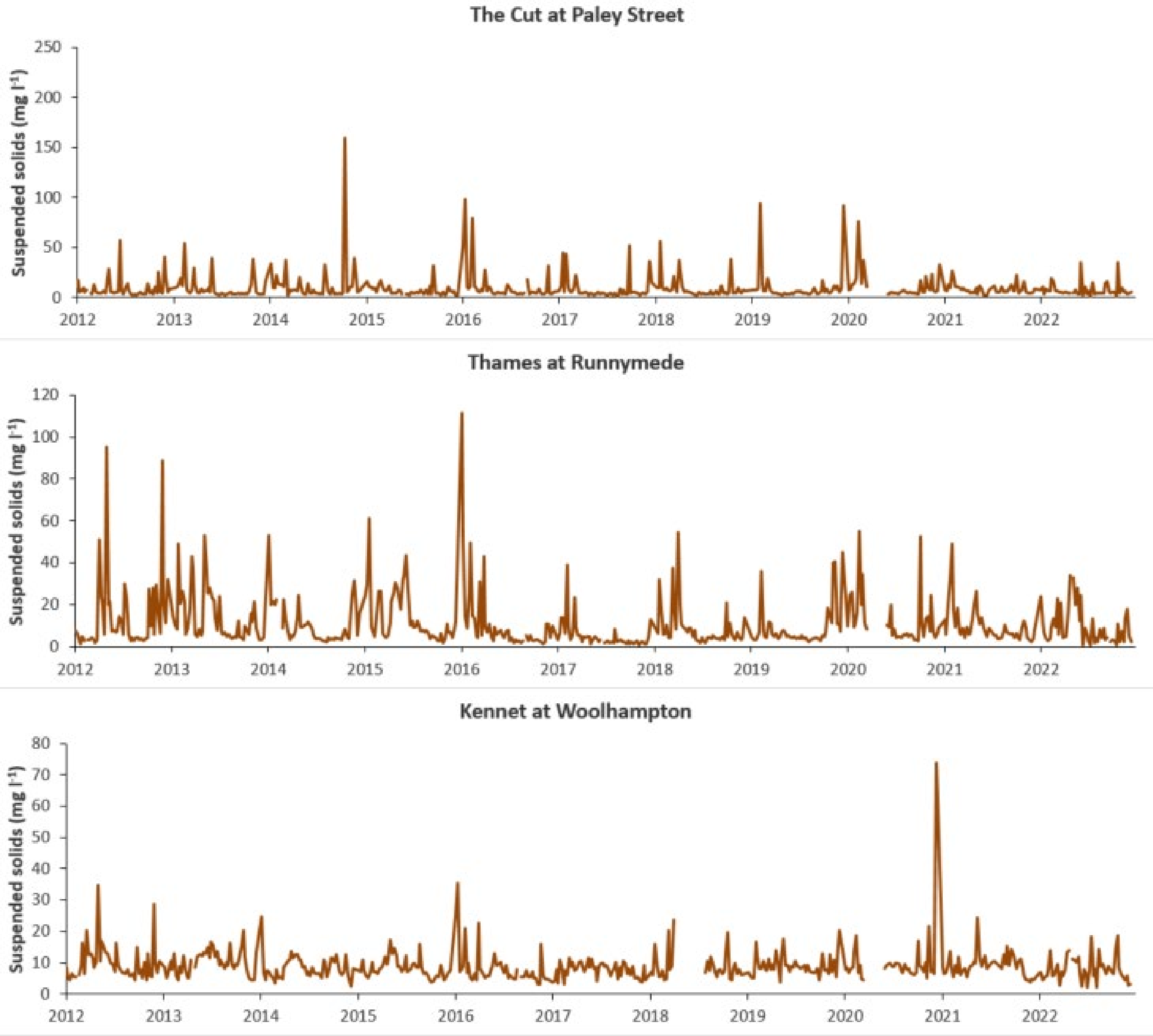

Figure 11. Weekly suspended solids concentrations at Thames case study locations

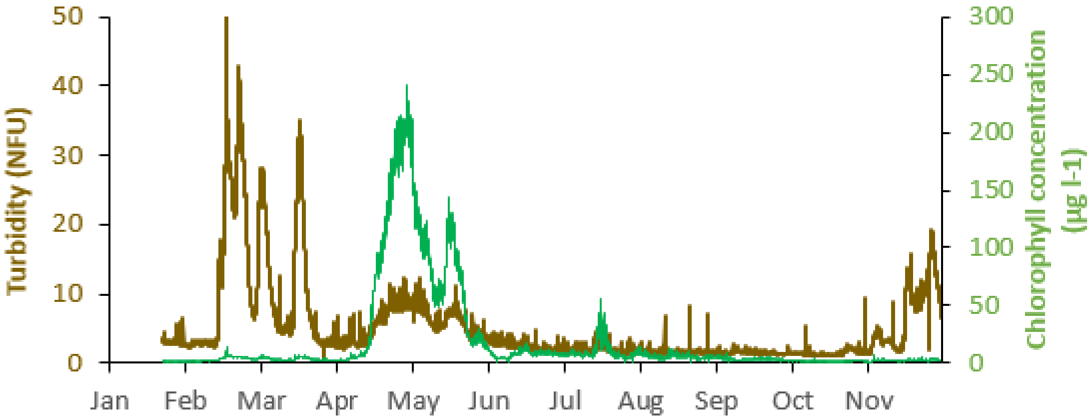

Figure 12. Hourly turbidity data from the River Thames at Taplow, Maidenhead (2022). (Data supplied by the Environment Agency)

The Thames high-frequency monitoring station at Taplow produced similar data (Figure 12), with the turbidity (a proxy for suspended solids) generally declining from April to October 2022, with a turbidity peak in April – May due to the increased phytoplankton load (as indicated by the chlorophyll concentration). The storms and higher flows in November resulted in higher turbidities, resulting from inwash of sediment and particulates from the catchment and re-suspension of bed sediments that had accumulated throughout the drought.

3.5. Dissolved oxygen

Many studies predict that droughts can result in low dissolved oxygen concentrations, due to increased water temperatures and subsequent reduction on oxygen solubility, high rates of bacterial respiration, and a lack of turbulence reducing re-aeration rates. In the case of the River Thames during the 2022 drought, the dissolved oxygen levels remained supersaturated (up to 190%) during periods of high chlorophyll concentrations (Figure 13). Although there was a large amplitude in the diurnal DO cycling, the oxygen concentrations remained highly oversaturated overnight. The River Thames is relatively long and slow flowing, and therefore is able to develop a significant phytoplankton biomass in its middle and lower reaches through the spring and summer periods. Shorter, faster-flowing rivers are less likely to have the algal and macrophytes biomass to produce similar rates of photosynthesis, but could still have high rates of microbial respiration, which may result in oxygen depletion during droughts.

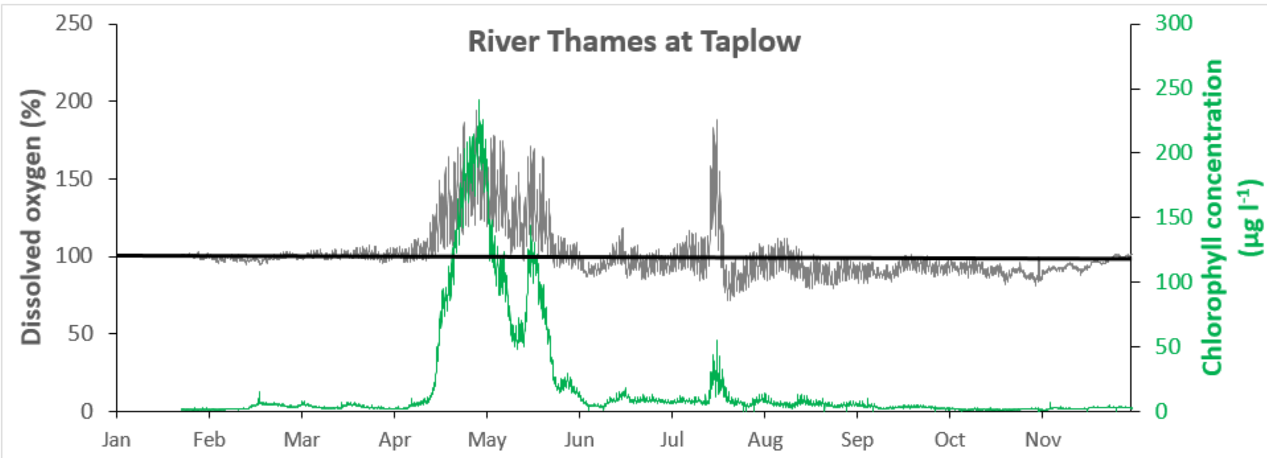

Figure 13. Hourly dissolved oxygen and chlorophyll concentrations in the River Thames at Taplow, through 2022 (Data supplied by the Environment Agency).

Previous studies have also predicted sags in DO concentration when phytoplankton blooms collapse. The Thames dataset demonstrates this, with DO concentrations falling below 100% at the end of the April-May and July chlorophyll peaks (Figure 13). DO concentrations fell to 70% in late July, when chlorophyll concentrations were low and water temperatures were at their highest, which supports many of the monitoring and modelling study predictions.

3.6. Chlorophyll

The 2022 drought produced the highest observed peak in chlorophyll concentration in the lower River Thames at Runnymede, although the chlorophyll bloom period was relatively short (7 weeks) (Figure 14). The bloom ended at the end of May 2022, and chlorophyll concentrations remained low (<10 µg l-1) throughout July and August, when the drought conditions were at their maximum. In contrast, the River Kennet and The Cut did not experience a chlorophyll bloom during 2022, and were unaffected by the drought.

Previous Thames studies have demonstrated that chlorophyll concentrations are directly related to diatom abundance (Moorhouse et al., 2018). These chlorophyll blooms have been shown to commence in the middle and lower Thames when flows are low, there is lots of sunlight and water temperatures are between 9 and 19oC (Bowes et al., 2016). It is likely that the chlorophyll bloom began and developed rapidly due to the relatively low flow (which increases diatom residence time) and dry sunny weather in the spring period of 2022.

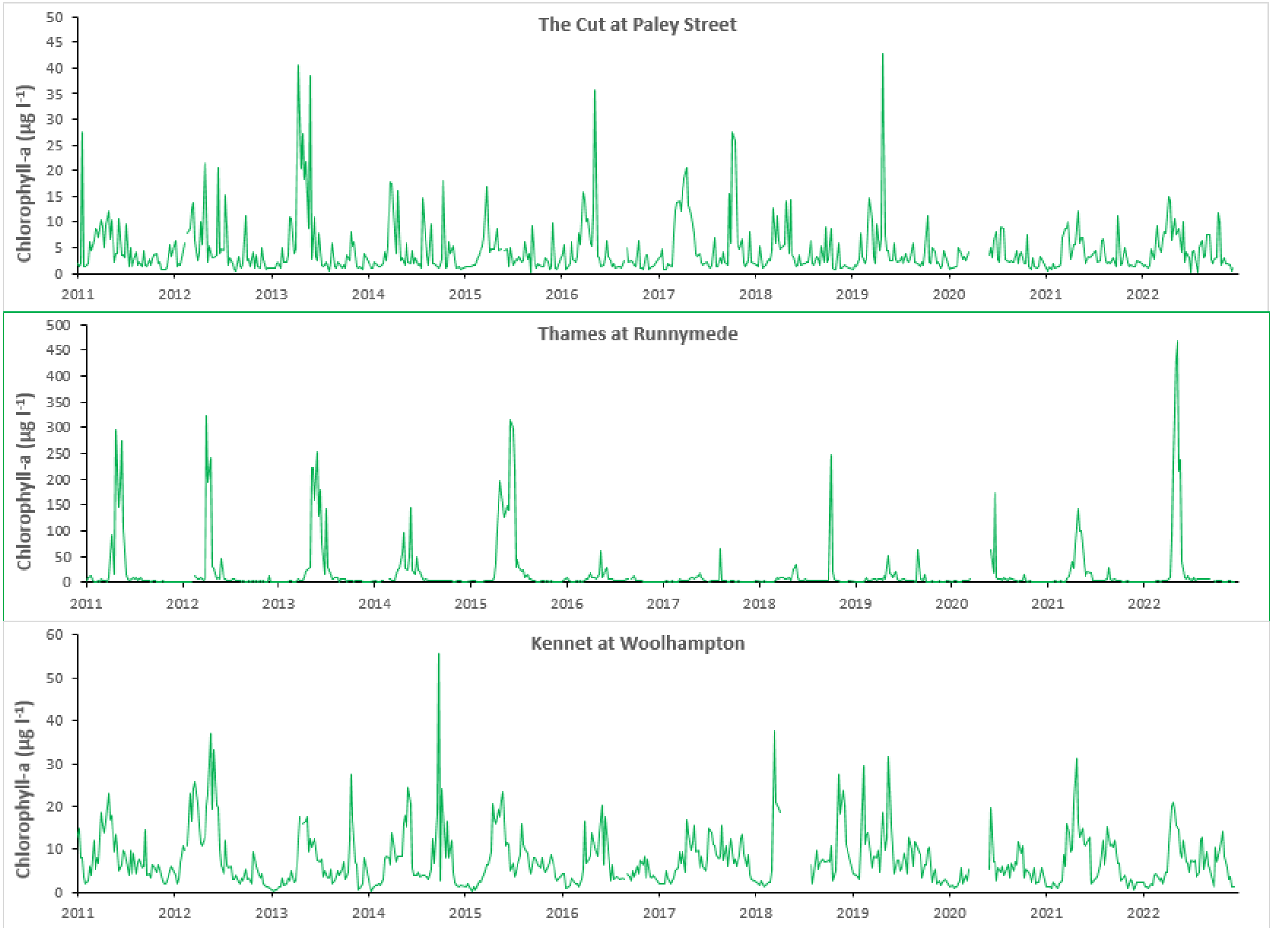

Figure 14. Weekly chlorophyll concentration data at Thames case study locations.

The increasing severity of the drought may have brought the chlorophyll bloom to an early termination, as diatoms decline rapidly at exceptionally-low flows, due to lack of turbulence resulting in deposition of the diatoms. This shows that the predicted increase in river chlorophyll blooms due to drought can happen. The low flow can result in more rapid growth rates, but can also terminate the bloom. It is likely that the timing of these blooms will change in the future, due to the predicted changes in hydrology and temperature. Consequently, the timing of the drought will have great bearing on algal biomass and community composition.

3.7. Phytoplankton

The UKCEH Thames Initiative monitoring incorporates weekly flow cytometry analysis for each study site. This provides cell abundances per ml for a range of 11 groups of phytoplankton (Read et al., 2014). For this case study, we present data for diatoms, nano-chlorophytes, pico-chlorophytes and total cyanobacteria groups. This phytoplankton abundance data indicates that there was no increase in phytoplankton population in either The Cut or the River Kennet throughout the 2022 drought period (

Figure 15, Figure 16). The data confirms that the high chlorophyll concentration in the Thames was due to high diatom cell abundance in April to May 2022, with the highest recorded diatom cell densities observed in the 11-year data record (Figure 17).

All sites showed a shift to pico-chlorophytes and cyanobacteria as the drought progressed, which is due to these groups being able to maintain their position in the water column under severe low-flow conditions, and also favouring higher water temperatures. Cyanobacteria as known to proliferate when water temperatures are high, and this data provides clear evidence for this, with cyanobacteria only proliferating when water temperatures were > 20oC (Figure 18). However the record water temperatures observed in the summer of 2022 in all three rivers did not produce record cyanobacteria cell abundances, as may have been expected. This demonstrates that other multiple parameters (light, nutrients, flow) are important in controlling cyanobacterial blooms, but increasing drought risk and water temperatures are likely to increase the risk of harmful cyanobacterial blooms in the future.

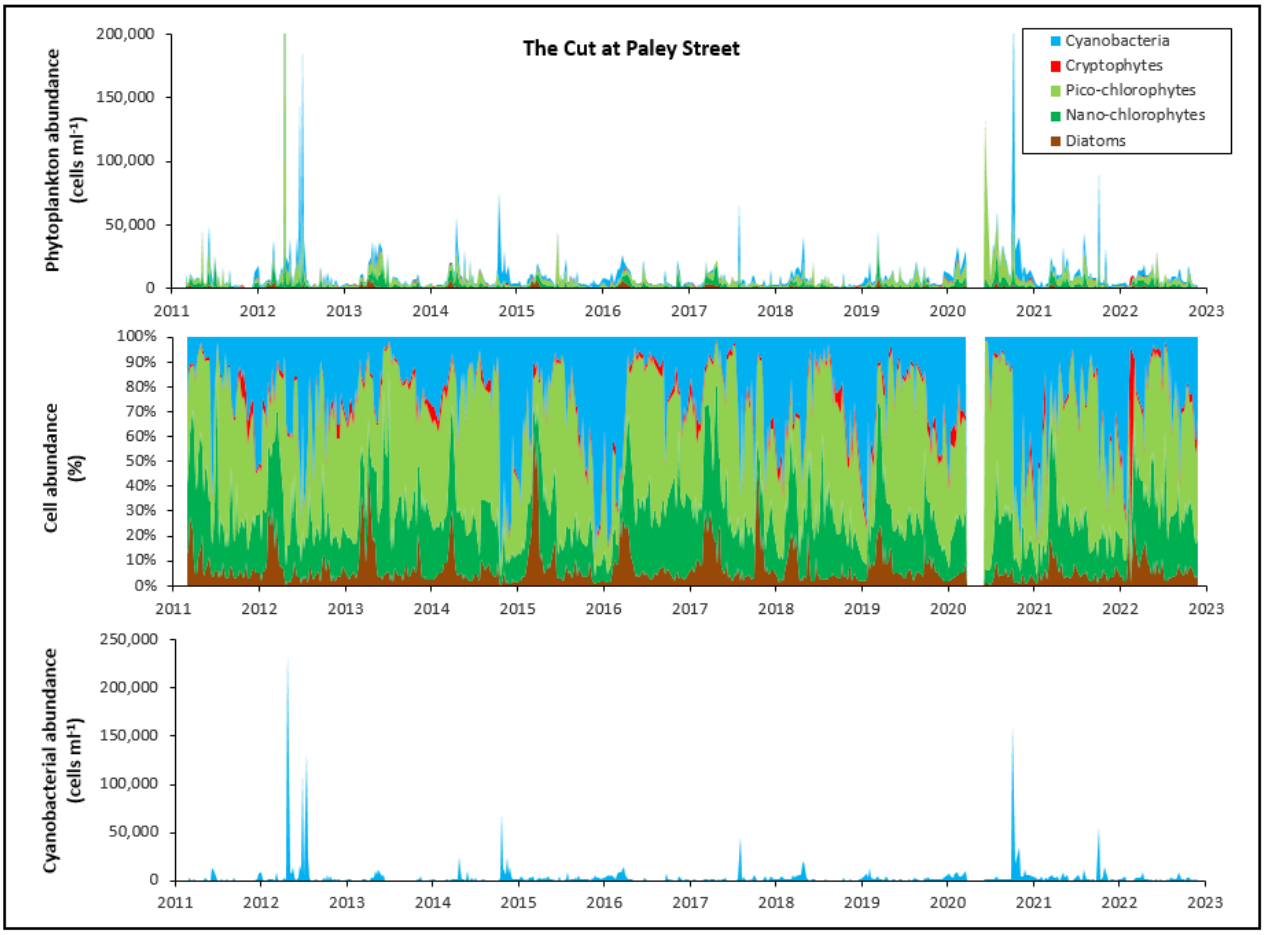

Figure 15. Weekly phytoplankton and cyanobacterial cell abundances in The Cut, analysed by Flow Cytometry.

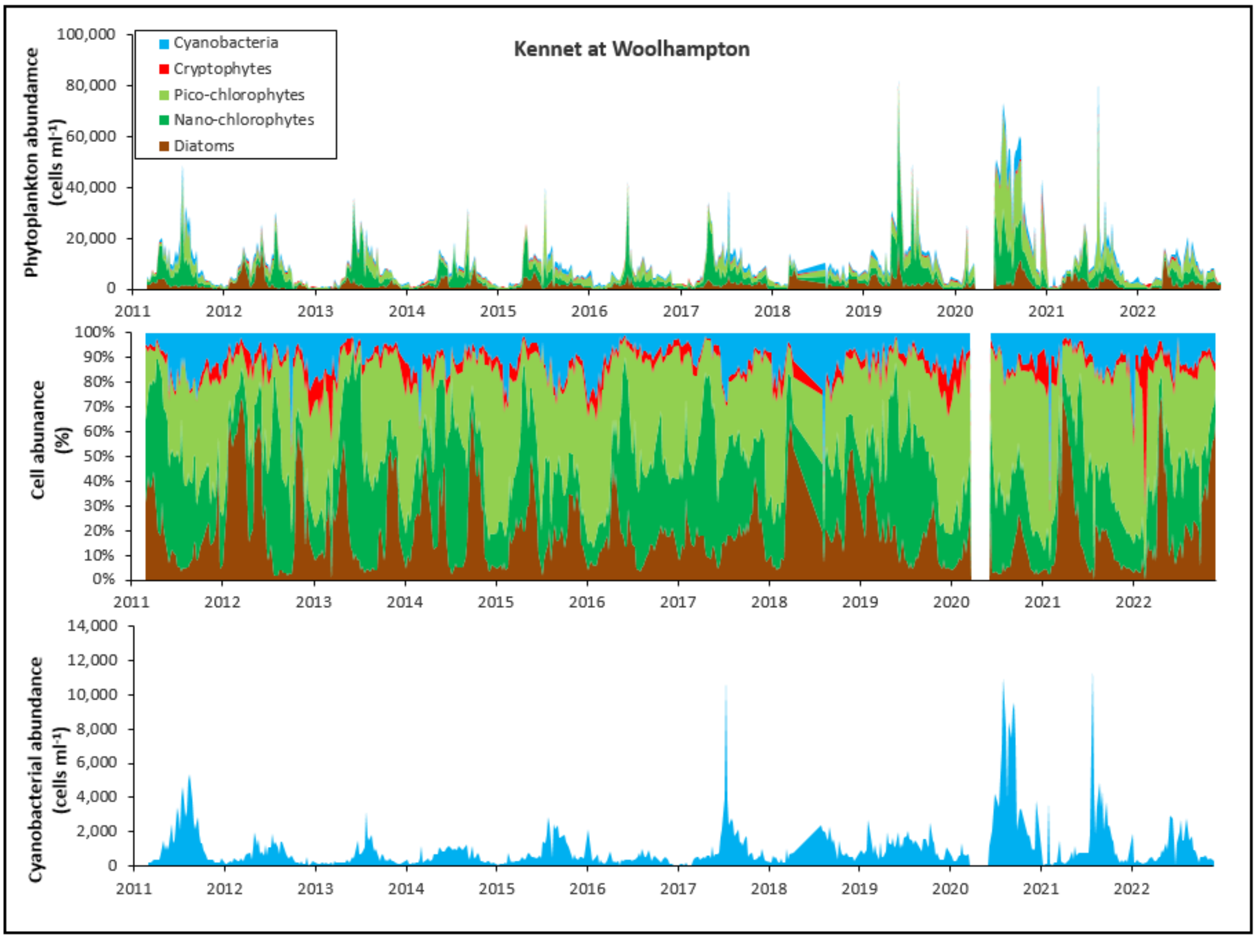

Figure 16. Weekly phytoplankton and cyanobacterial cell abundances in the River Kennet, analysed by Flow Cytometry.

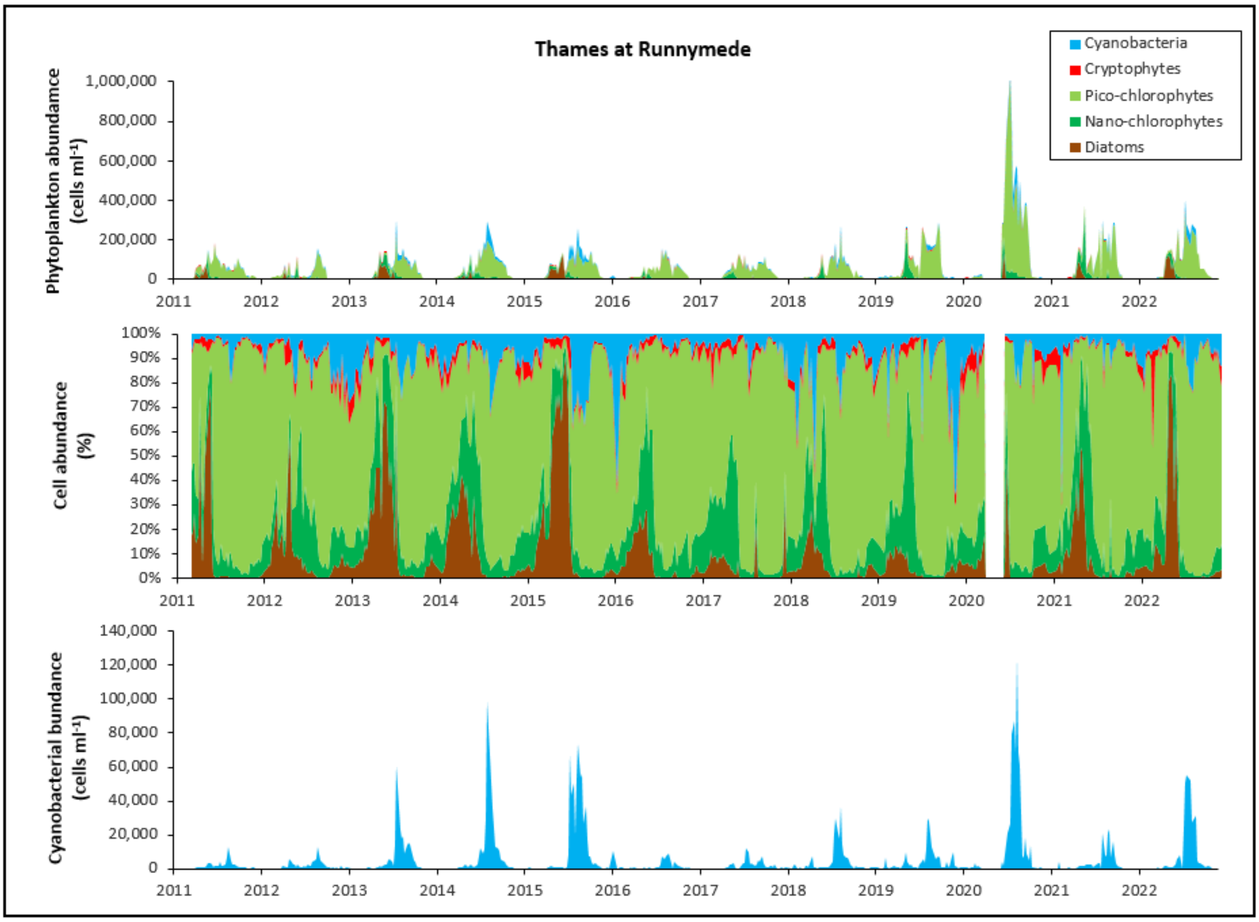

Figure 17. Weekly phytoplankton and cyanobacterial cell abundances in the River Thames, analysed by Flow Cytometry.

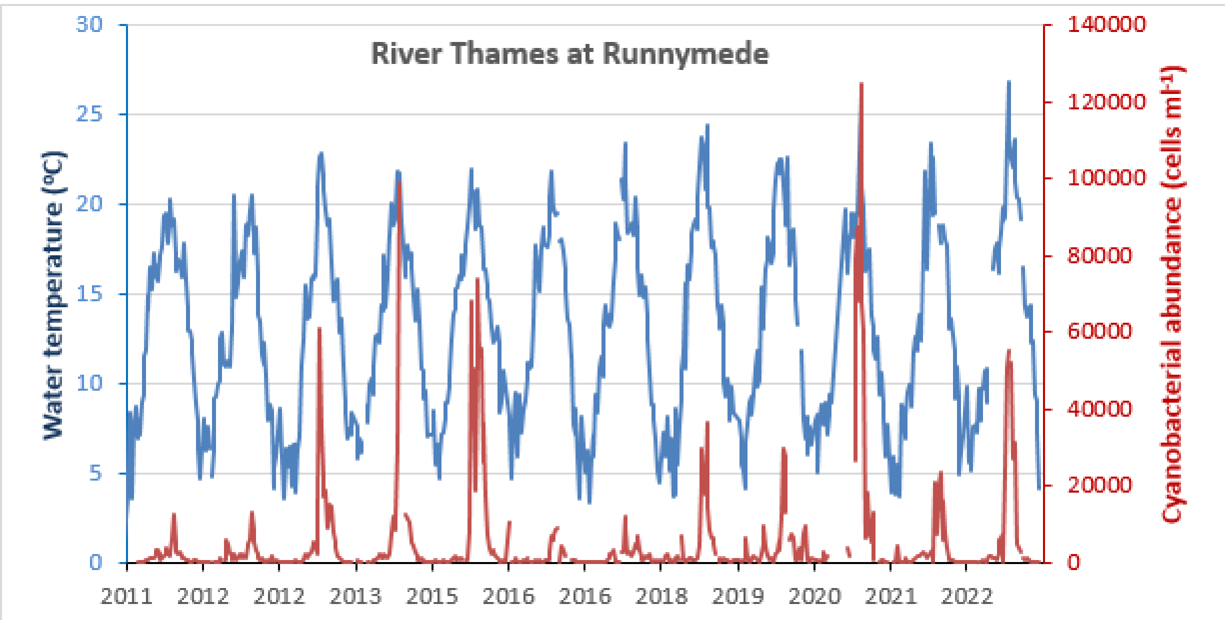

Figure 18. Relationship between water temperature and cyanobacterial cell abundance.

4. Remaining research gaps

It is clear that droughts can directly impact a wide range of physical, chemical and biological water quality parameters. There is an urgent need to increase the spatial and temporal scale of long term monitoring to allow the research community to gain new insights into drought impacts when they occur. The NERC Floods and Droughts Research Infrastructure that is currently being implemented could provide the data required when droughts next affect the UK.

One of the commonly-predicted changes in rivers and lakes in the future is that there will be an increase in algal and cyanobacterial bloom risk. To capture the underpinning data required to understand this, there is a need to increase phytoplankton monitoring. High-frequency chlorophyll probes are now reliable and available, and should be deployed in our larger UK rivers where algal blooms can be a problem. Additional phytoplankton community composition data is also vital, using flow cytometry or specialist fluorimeters that can quantify the concentrations of different photosynthetic pigments. One of the major reasons that there is concern over the potential increase in cyanobacterial populations during periods of drought and heat waves is that they can produce neurotoxins that can poison livestock and pets, and can cause problems for drinking water supplies. Monitoring these neurotoxin compounds in at-risk waterbodies is a really important step to develop understanding of the conditions in which these cyanobacteria proliferate, and what causes them to produce these toxins.

Environmental flows (e-flows) indicate both the quantity and quality of water required to sustain river and water-dependent ecosystems. Although the definition of e-flows incorporates water quality, almost all approaches have focused on water quantity and flow regime, with few covering water quality. Water quality experts have set many standards which are not formally linked to e-flows. Yet, most rivers are exposed to multiple stressors from human activities, of which water quality is one, which are difficult to disentangle; environmental flow assessments need to consider stressor interactions (Laize et al., 2021). There is a need to investigate the impact of droughts on WQ and river ecology together rather than separately.

The Thames case study has shown that the 2022 drought has had varying impacts on different waterbody types. More work is urgently needed to assess which waterbody types are most vulnerable to drought conditions, and which are most resilient. This could provide vital insights into how rivers and lakes could be best managed to increase their resilience to future climate change.

There are many mitigation options available to catchment managers to reduce drought impacts, such as reducing water use, flow-support schemes, cooling of rivers by riparian tree planting, improved wastewater treatment etc. It is important to try to capture how these interventions reduce the impact of drought on water quality and eutrophication risk, to provide the evidence and support modelling activities.

5. Opportunities and recommendations

Targeted and comprehensive monitoring needs to be in place to capture the impacts of future droughts, from pre-drought conditions through to the recovery period. Commencing monitoring once a drought has been declared is too late. Hopefully the NERC Floods and Droughts Research Infrastructure programme will provide this platform over the coming decades for a range of typical UK catchments.

Capturing full biogeochemical datasets through future droughts is vital, as it provides the much-needed evidence to determine if the predictions of potential drought impacts discussed in this report do actually occur, and how widespread and severe these impacts are.

It is vital to establish the river and lake typologies whose water quality and ecology are most likely to be affected by droughts, as this will allow the UK to target mitigation resources. Important lessons can be learnt from identifying drought-resilient catchments, as this can inform how to increase resilience in vulnerable catchments through land-use change, water quality improvements and river restoration.

Existing water quality data sets from regulators and the academic community through the summer of 2022 and previous drought periods should be compiled and interrogated to determine levels of impacts and the spatial pattern in drought impacts.

Finally, appropriate biogeochemical monitoring, alongside system understanding and modelling, could provide early warning systems to alert the presence / development of poor water quality, low DO, and the development of algal and cyanobacterial blooms. This would be of great value to the Environment Agency, allowing them to protect aquatic ecosystems and human health during periods of drought stress, and be extremely useful to Water Companies and human water supply.

References

Bieroza M, Bergstrom L, Ulen B, Djodjic F, Tonderski K, Heeb A, et al. Hydrologic Extremes and Legacy Sources Can Override Efforts to Mitigate Nutrient and Sediment Losses at the Catchment Scale. J Environ Qual 2019; 48: 1314-1324.

Booker DJ, Whitehead AL. River water temperatures are higher during lower flows after accounting for meteorological variability. River Research and Applications 2022; 38: 3-22.

Bowes MJ, Armstrong LK, Harman SA, Wickham HD, Nicholls DJE, Scarlett PM, et al. Weekly water quality monitoring data for the River Thames (UK) and its major tributaries (2009-2013): the Thames Initiative research platform. Earth System Science Data 2018; 10: 1637-1653.

Bowes MJ, Gozzard E, Johnson AC, Scarlett PM, Roberts C, Read DS, et al. Spatial and temporal changes in chlorophyll-a concentrations in the River Thames basin, UK: are phosphorus concentrations beginning to limit phytoplankton biomass? Sci Total Environ 2012; 426: 45-55.

Bowes MJ, House WA. Phosphorus and dissolved silicon dynamics in the River Swale catchment, UK: a mass-balance approach. Hydrological Processes 2001; 15: 261-280.

Bowes MJ, Jarvie HP, Halliday SJ, Skeffington RA, Wade AJ, Loewenthal M, et al. Characterising phosphorus and nitrate inputs to a rural river using high-frequency concentration-flow relationships. Sci Total Environ 2015a; 511: 608-20.

Bowes MJ, Jarvie HP, Halliday SJ, Skeffington RA, Wade AJ, Loewenthal M, et al. Characterising phosphorus and nitrate inputs to a rural river using high-frequency concentration-flow relationships. Science of the Total Environment 2015b; 511: 608-620.

Bowes MJ, Jarvie HP, Naden PS, Old GH, Scarlett PM, Roberts C, et al. Identifying priorities for nutrient mitigation using river concentration–flow relationships: The Thames basin, UK. Journal of Hydrology 2014; 517: 1-12.

Bowes MJ, Loewenthal M, Read DS, Hutchins MG, Prudhomme C, Armstrong LK, et al. Identifying multiple stressor controls on phytoplankton dynamics in the River Thames (UK) using high-frequency water quality data. Sci Total Environ 2016; 569-570: 1489-1499.

Bowes MJ, Smith JT, Jarvie HP, Neal C. Modelling of phosphorus inputs to rivers from diffuse and point sources. Sci Total Environ 2008; 395: 125-38.

Burt TP, Worrall F, Howden NJK, Anderson MG. Shifts in discharge-concentration relationships as a small catchment recover from severe drought. Hydrological Processes 2015; 29: 498-507.

Bussi G, Janes V, Whitehead PG, Dadson SJ, Holman IP. Dynamic response of land use and river nutrient concentration to long-term climatic changes. Sci Total Environ 2017; 590-591: 818-831.

Bussi G, Whitehead PG, Bowes MJ, Read DS, Prudhomme C, Dadson SJ. Impacts of climate change, land-use change and phosphorus reduction on phytoplankton in the River Thames (UK). Sci Total Environ 2016; 572: 1507-1519.

Bussi G, Whitehead PG, Gutierrez-Canovas C, Ledesma JLJ, Ormerod SJ, Couture RM. Modelling the effects of climate and land-use change on the hydrochemistry and ecology of the River Wye (Wales). Sci Total Environ 2018; 627: 733-743.

Byrne P, Onnis P, Runkel RL, Frau I, Lynch SFL, Edwards P. Critical Shifts in Trace Metal Transport and Remediation Performance under Future Low River Flows. Environ Sci Technol 2020; 54: 15742-15750.

Carvalho L, Miller nee Ferguson CA, Scott EM, Codd GA, Davies PS, Tyler AN. Cyanobacterial blooms: statistical models describing risk factors for national-scale lake assessment and lake management. Sci Total Environ 2011; 409: 5353-8.

Charlton MB, Bowes MJ, Hutchins MG, Orr HG, Soley R, Davison P. Mapping eutrophication risk from climate change: Future phosphorus concentrations in English rivers. Sci Total Environ 2018; 613-614: 1510-1526.

Chessman BC. Relationships between lotic macroinvertebrate traits and responses to extreme drought. Freshwater Biology 2015; 60: 50-63.

Costa D, Sutter C, Shepherd A, Jarvie H, Wilson H, Elliott J, et al. Impact of climate change on catchment nutrient dynamics: insights from around the world. Environmental Reviews 2022.

Desortova B, Puncochar P. Variability of phytoplankton biomass in a lowland river: Response to climate conditions. Limnologica 2011; 41: 160-166.

Dollar E, Edwards F, Stratford C, May L, Biggs J, Laize C, et al. Monitoring and Assessment of Environmental Impacts of Droughts: Literature synthesis, Bristol, UK, 2012.

Elliott JA. The seasonal sensitivity of Cyanobacteria and other phytoplankton to changes in flushing rate and water temperature. Global Change Biology 2010; 16: 864-876.

Environment Agency. Climate change and eutrophication risk in English rivers. Chief Scientist's Group report. Environment Agency, Bristol, 2016.

Forber KJ, Withers PJA, Ockenden MC, Haygarth PM. The Phosphorus Transfer Continuum: A Framework for Exploring Effects of Climate Change. Agricultural & Environmental Letters 2018; 3: 180036.

Gomez-Gener L, Lupon A, Laudon H, Sponseller RA. Drought alters the biogeochemistry of boreal stream networks. Nat Commun 2020; 11: 1795.

Halliday SJ, Skeffington RA, Wade AJ, Bowes MJ, Gozzard E, Newman JR, et al. High-frequency water quality monitoring in an urban catchment: hydrochemical dynamics, primary production and implications for the Water Framework Directive. Hydrological Processes 2015; 29: 3388-3407.

Havens KE, Ji GH, Beaver JR, Fulton RS, Teacher CE. Dynamics of cyanobacteria blooms are linked to the hydrology of shallow Florida lakes and provide insight into possible impacts of climate change. Hydrobiologia 2019; 829: 43-59.

Hellwig J, Stahl K, Lange J. Patterns in the linkage of water quantity and quality during low-flows. Hydrological Processes 2017; 31: 4195-4205.

Hering D, Haidekker A, Schmidt-Kloiber A, Barker T, Buisson L, Graf W, et al. Monitoring the Responses of Freshwater Ecosystems to Climate Change. Climate Change Impacts on Freshwater Ecosystems, 2010, pp. 84-118.

Hilton J, O'Hare M, Bowes MJ, Jones JI. How green is my river? A new paradigm of eutrophication in rivers. Science of the Total Environment 2006; 365: 66-83.

Hosen JD, Aho KS, Appling AP, Creech EC, Fair JH, Hall RO, et al. Enhancement of primary production during drought in a temperate watershed is greater in larger rivers than headwater streams. Limnology and Oceanography 2019; 64: 1458-1472.

Hrdinka T, Novicky O, Hanslik E, Rieder M. Possible impacts of floods and droughts on water quality. Journal of Hydro-Environment Research 2012; 6: 145-150.

Hutchins MG, Hitt OE. Sensitivity of river eutrophication to multiple stressors illustrated using graphical summaries of physics-based river water quality model simulations. Journal of Hydrology 2019; 577: 11.

Hutchins MG, Qu YM, Charlton MB. Successful modelling of river dissolved oxygen dynamics requires knowledge of stream channel environments. Journal of Hydrology 2021; 603: 126991.

Jarvie HP, Pallett DW, Schafer SM, Macrae ML, Bowes MJ, Farrand P, et al. Biogeochemical and climate drivers of wetland phosphorus and nitrogen release: Implications for nutrient legacies and eutrophication risk. J Environ Qual 2020; 49: 1703-1716.

Ji GH, Havens KE, Beaver JR, Fulton RS. Response of Zooplankton to Climate Variability: Droughts Create a Perfect Storm for Cladocerans in Shallow Eutrophic Lakes. Water 2017; 9: 764.

Jin L, Whitehead PG, Futter MN, Lu ZL. Modelling the impacts of climate change on flow and nitrate in the River Thames: assessing potential adaptation strategies. Hydrology Research 2012; 43: 902-916.

Johnson AC, Acreman MC, Dunbar MJ, Feist SW, Giacomello AM, Gozlan RE, et al. The British river of the future: How climate change and human activity might affect two contrasting river ecosystems in England. Science of the Total Environment 2009; 407: 4787-4798.

Jones E, van Vliet MTH. Drought impacts on river salinity in the southern US: Implications for water scarcity. Sci Total Environ 2018; 644: 844-853.

Kamjunke N, Beckers LM, Herzsprung P, von Tumpling W, Lechtenfeld O, Tittel J, et al. Lagrangian profiles of riverine autotrophy, organic matter transformation, and micropollutants at extreme drought. Sci Total Environ 2022; 828: 154243.

Laize C, Acreman MC, Azevedo D, Bloomfield JP, Bulcock A, Cross P, et al. Review of Approaches for Determining the Environmental Water Need: Synthesis. Report to Defra, 2021, pp. 21.

Lake PS. Water Bodies, Catchments and the Abiotic Effects of Drought. Drought and Aquatic Ecosystems: Effects and Responses, 2011, pp. 68-99.

Lisboa MS, Schneider RL, Sullivan PJ, Walter MT. Drought and post-drought rain effect on stream phosphorus and other nutrient losses in the Northeastern USA. Journal of Hydrology-Regional Studies 2020; 28: 100672.

Loecke TD, Burgin AJ, Riveros-Iregui DA, Ward AS, Thomas SA, Davis CA, et al. Weather whiplash in agricultural regions drives deterioration of water quality. Biogeochemistry 2017; 133: 7-15.

Martin N, McEachern P, Yu T, Zhu DZ. Model development for prediction and mitigation of dissolved oxygen sags in the Athabasca River, Canada. Sci Total Environ 2013; 443: 403-12.

McGrane SJ, Hutchins MG, Miller JD, Bussi G, Kjeldsen TR, Loewenthal M. During a winter of storms in a small UK catchment, hydrology and water quality responses follow a clear rural-urban gradient. Journal of Hydrology 2017; 545: 463-477.

Miller JD, Hutchins M. The impacts of urbanisation and climate change on urban flooding and urban water quality: A review of the evidence concerning the United Kingdom. Journal of Hydrology-Regional Studies 2017; 12: 345-362.

Minaudo C, Abonyi A, Leitao M, Lancon AM, Floury M, Descy JP, et al. Long-term impacts of nutrient control, climate change, and invasive clams on phytoplankton and cyanobacteria biomass in a large temperate river. Sci Total Environ 2021; 756: 144074.

Moorhouse HL, Read DS, McGowan S, Wagner M, Roberts C, Armstrong LK, et al. Characterisation of a major phytoplankton bloom in the River Thames (UK) using flow cytometry and high performance liquid chromatography. Sci Total Environ 2018; 624: 366-376.

Mosley LM. Drought impacts on the water quality of freshwater systems; review and integration. Earth-Science Reviews 2015; 140: 203-214.

Mosley LM, Zammit B, Jolley AM, Barnett L. Acidification of lake water due to drought. Journal of Hydrology 2014; 511: 484-493.

Naden PS, Collins AL, Bowes MJ, Edwards F, al e. Developing a field tool kit for ecological targeting of agricultural diffuse pollution mitigation measures. Department for Environment, Food and Rural Affairs, 2015.

O'Hare MT, Baattrup-Pedersen A, Baumgarte I, Freeman A, Gunn IDM, Lazar AN, et al. Responses of Aquatic Plants to Eutrophication in Rivers: A Revised Conceptual Model. Front Plant Sci 2018; 9: 451.

Ockenden MC, Deasy CE, Benskin CMH, Beven KJ, Burke S, Collins AL, et al. Changing climate and nutrient transfers: Evidence from high temporal resolution concentration-flow dynamics in headwater catchments. Sci Total Environ 2016; 548-549: 325-339.

Outram FN, Cooper RJ, Sunnenberg G, Hiscock KM, Lovett AA. Antecedent conditions, hydrological connectivity and anthropogenic inputs: Factors affecting nitrate and phosphorus transfers to agricultural headwater streams. Science of the Total Environment 2016; 545: 184-199.

Paerl HW, Huisman J. Climate. Blooms like it hot. Science 2008; 320: 57-8.

Parr LB, Mason CF. Long-term trends in water quality and their impact on macroinvertebrate assemblages in eutrophic lowland rivers. Water Res 2003; 37: 2969-79.

Pathak D, Hutchins M, Brown L, Loewenthal M, Scarlett P, Armstrong L, et al. Hourly Prediction of Phytoplankton Biomass and Its Environmental Controls in Lowland Rivers. Water Resources Research 2021; 57: e2020WR028773.

Piano E, Falasco E, Bona F. Mediterranean rivers: Consequences of water scarcity on benthic algal chlorophyll a content. Journal of Limnology 2017; 76: 39-48.

Prudhomme C, Dadson S, Morris D, Williamson J, Goodsell G, Crooks S, et al. Future Flows Climate: an ensemble of 1-km climate change projections for hydrological application in Great Britain. Earth System Science Data 2012; 4: 143-148.

Pulley S, Foster I, Antunes P. The dynamics of sediment-associated contaminants over a transition from drought to multiple flood events in a lowland UK catchment. Hydrological Processes 2016; 30: 704-719.

Raimonet M, Thieu V, Silvestre M, Oudin L, Rabouille C, Vautard R, et al. Landward Perspective of Coastal Eutrophication Potential Under Future Climate Change: The Seine River Case (France). Frontiers in Marine Science 2018; 5.

Read DS, Bowes MJ, Newbold LK, Whiteley AS. Weekly flow cytometric analysis of riverine phytoplankton to determine seasonal bloom dynamics. Environ Sci Process Impacts 2014; 16: 594-603.

Reichwaldt ES, Ghadouani A. Effects of rainfall patterns on toxic cyanobacterial blooms in a changing climate: between simplistic scenarios and complex dynamics. Water Res 2012; 46: 1372-93.

Reynolds CS. The ecology of phytoplankton: Cambridge University Press, 2006.

van Vliet MTH, Zwolsman JJG. Impact of summer droughts on the water quality of the Meuse river. Journal of Hydrology 2008; 353: 1-17.

Vilmin L, Flipo N, Escoffier N, Rocher V, Groleau A. Carbon fate in a large temperate human-impacted river system: Focus on benthic dynamics. Global Biogeochemical Cycles 2016; 30: 1086-1104.

Wade AJ, Skeffington RA, Couture RM, Lampa ME, Groot S, Halliday SJ, et al. Land Use Change to Reduce Freshwater Nitrogen and Phosphorus will Be Effective Even with Projected Climate Change. Water 2022; 14.

Warren M, Dunbar MJ, Smith C. River flow as a determinant of salmonid distribution and abundance: a review. Environmental Biology of Fishes 2015; 98: 1695-1717.

Watkins L, Sullivan PJ, Walter MT. A case study investigating temporal factors that influence microplastic concentration in streams under different treatment regimes. Environ Sci Pollut Res Int 2019; 26: 21797-21807.

Waylett AJ, Hutchins MG, Johnson AC, Bowes MJ, Loewenthal M. Physico-chemical factors alone cannot simulate phytoplankton behaviour in a lowland river. Journal of Hydrology 2013; 497: 223-233.

Whelan MJ, Linstead C, Worrall F, Ormerod SJ, Durance I, Johnson AC, et al. Is water quality in British rivers "better than at any time since the end of the Industrial Revolution"? Sci Total Environ 2022; 843: 157014.

Whitehead PG, Wilby RL, Battarbee RW, Kernan M, Wade AJ. A review of the potential impacts of climate change on surface water quality. Hydrological Sciences Journal-Journal Des Sciences Hydrologiques 2009; 54: 101-123.

Whitehead PG, Wilson EJ, Butterfield D. A semi-distributed Integrated Nitrogen model for multiple source assessment in Catchments (INCA): Part I - model structure and process equations. Science of the Total Environment 1998; 210: 547-558.

Zhi W, Feng D, Tsai WP, Sterle G, Harpold A, Shen C, et al. From Hydrometeorology to River Water Quality: Can a Deep Learning Model Predict Dissolved Oxygen at the Continental Scale? Environ Sci Technol 2021; 55: 2357-2368.

F: Drought-induced river water temperature dynamics in the United Kingdom

J. C. White, K. Khamis and D. M. Hannah

School of Geography, Earth & Environmental Sciences, University of Birmingham, Birmingham, UK

Overview

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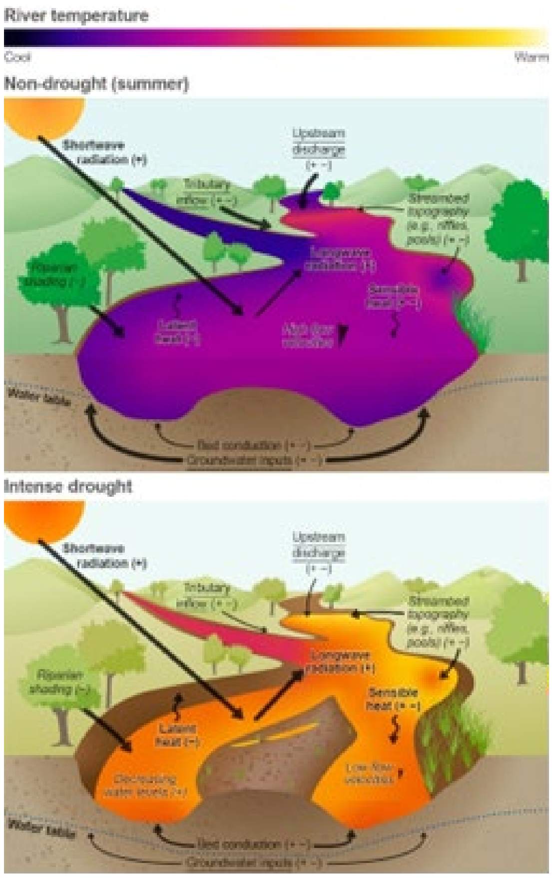
High river water temperature extremes have been widely reported during droughts, with major ecological and management implications. However, the extent to which different meteorological (e.g., high air temperatures) and hydrological processes (e.g., shifting flow hydraulics and water source contributions) interact during droughts to govern river temperature dynamics, and how this varies between environmental contexts, remains poorly understood. In this context, we review mechanisms operating across three ‘process sets’ governing river temperature extremes during droughts: (i) **energy flux dynamics** governing heat transfer processes; (ii) **reach-scale habitats** mediating heat flux effects, including hydraulic (e.g., residence time) and physical conditions (e.g., riparian vegetation coverages, wetted perimeters); (iii) **water source contributions** (surface water and groundwater) as advective heat and water flow controls.



Key findings

We identified high river temperature extremes during nationally iconic droughts (e.g., 1976, 1995, 2018). However, wider drought-river temperature associations have been inconsistent, reflecting the sensitivity of water temperature to environmental controls beyond low-flows experienced during droughts.

UK studies examining the mechanisms governing drought-induced river temperature dynamics have largely stemmed from studies with broader focuses (e.g., annual thermal dynamics). Such evidence is therefore disjointed and spatially biased (a focus on SW England and NE Scotland). Notwithstanding, we have conceptualised key changes to thermal drivers during droughts based on a process-based understanding.

Solar radiation represents the primary heat input elevating river temperature during droughts. Shading and groundwater inputs can offset radiative heat inputs, but their cooling influences varies markedly within and between river catchments. Enhanced modelling capabilities are providing greater evidence-based guidance for riparian planting management practices. Specifically, research has indicated that afforesting slower flowing headwater systems can be most effective in reducing high river temperature extremes during low-flow conditions.

Recommendations

Processes – Our scientific thinking on river temperature extremes during drought (and heatwaves) needs to be re-conceptualised so that **energy flux dynamics, reach-scale habitat** (hydraulic and riparian) influences and water source contributions (advective flows and heat) are considered and their interactions quantified.

Data - Large-scale, high frequency river temperature monitoring networks spanning different environmental gradients are required to more accurately model river temperature dynamics during future droughts and at unmonitored sites. Drought effects of different **temperature drivers across multiple process sets** are also needed across **various river environments** to better support model predictions.

Models - Hydrological information that better characterise drought conditions should be more widely incorporated within river temperature models that are often based on climate and landscape controls.

Decision making - An evidence-based approach that utilises river temperature data and models is required to **better inform management interventions** capable of mitigating high temperature extremes.

Abstract

High river water temperature (Tw) extremes have been widely reported during drought conditions as extreme low-flows often coincide with high atmospheric energy inputs. This has significant ramifications for freshwater ecosystem health and sustainable river management practices worldwide. However, the extent to which different meteorological (e.g., high air temperatures) and hydrological processes (e.g., shifting flow hydraulics and water source contributions) interact during droughts to govern Tw dynamics, and how this varies between environmental contexts, remains poorly understood. In this context, we review evidence on the mechanisms underpinning Tw dynamics during droughts across the UK, and supported this with study findings from other temperate, maritime environments globally. From the literature reviewed and raw data presented, we evidence that Tw spikes have widely occurred during nationally iconic droughts (e.g., 1976, 1995, 2018). However, such trends were inconsistent, reflecting the sensitivity of drought-induced Tw dynamics to variable environmental controls. To better understand these interacting processes and provide more generalisable knowledge, we reviewed mechanisms operating across three ‘process sets’ governing Tw extremes during droughts: (i) energy flux dynamics; (ii) reach-scale habitat conditions (e.g., hydraulic and riparian habitat conditions); (iii) and water source contributions (i.e., surface water and groundwater). After reviewing natural and anthropogenic influences of Tw dynamics across these process sets (‘What do we know?’) and highlighting fundamental knowledge gaps (‘What don’t we know?’) with this, we outline how new scientific evidence can underpin modelling efforts predicting Tw to present-day and future drought conditions. From a management perspective, we recommend that: our scientific thinking around river temperature extremes during drought (and heatwaves) needs to be re-conceptualised so that energy flux dynamics, reach-scale habitat (hydraulic and riparian) influences and water source contributions (advective flows and heat) are considered and their interactions quantified (processes); large-scale, high frequency river temperature monitoring networks spanning different environmental gradients are implemented more widely to more accurately model river temperature dynamics during future droughts and at unmonitored sites, and that drought effects on different Tw drivers spanning multiple process sets are required across various river environments to better support model predictions (data); hydraulic and hydrological information that better characterise drought conditions should be more widely incorporated within Tw models (modelling); and evidence-based approaches that utilise river temperature data and models are more widely utilised to better inform management interventions capable of mitigating high temperature extremes.

1. Introduction

River temperature is widely recognised as a ‘master’ water quality variable in lotic environments due to its fundamental importance for freshwater ecosystems (Olden and Naiman, 2010; Bonacina et al., 2023) and society (Ficklin et al., 2023). River water temperatures (‘Tw’ herein) directly control organism survival and behaviours, including their physiological responses to temperature and thermal signals triggering behavioural responses (e.g., migratory cues; Elliot and Elliot, 2010; Bonacina et al., 2023). Tw also governs the density and solubility of gases and chemical reaction rates that shape ecosystem functions like nutrient turnover and uptake (Bell et al., 2021) and respiration and metabolism (Arias Font et al., 2021). It directly controls dissolved oxygen concentrations essential for all aerobic organisms inhabiting freshwater environments (Verberk et al., 2016).

Droughts are broadly defined as an extreme lack of water relative to 'normal' conditions, and have been associated with high Tw extremes (van Vliet et al., 2011; Liu et al., 2020). Droughts can be characterised from meteorological, hydrological, agricultural / soil moisture, socio-economic and ecological perspectives (van Loon et al., 2016; White et al., 2022). Tw dynamics during drought are largely driven by atmospheric water deficits (meteorological) that reduce surface water and potentially groundwater availability (hydrological; see Fig. 1), and reduce thermal buffering capacity to radiative forcings. Tw is highly correlated to elevated air temperature and the accompanying high solar radiative forcings, including heatwaves (van Vliet et al., 2011) - defined as prolonged periods of unusually high air temperatures (normally spanning at least three consecutive days - Beckett and Sanderson, 2022). Hot and dry weather conditions often co-occur as reduced surface water volumes limit evaporative cooling and dry soils instigate atmospheric heat transfers (AghaKouchak et al., 2020), and thus have significant implications for Tw as these synergistic forcings operate simultaneously. Compound (co-occurring) drought-heatwaves particularly threaten freshwater ecosystems through Tw extremes, but they have been sparsely explored globally (AghaKouchak et al., 2020).

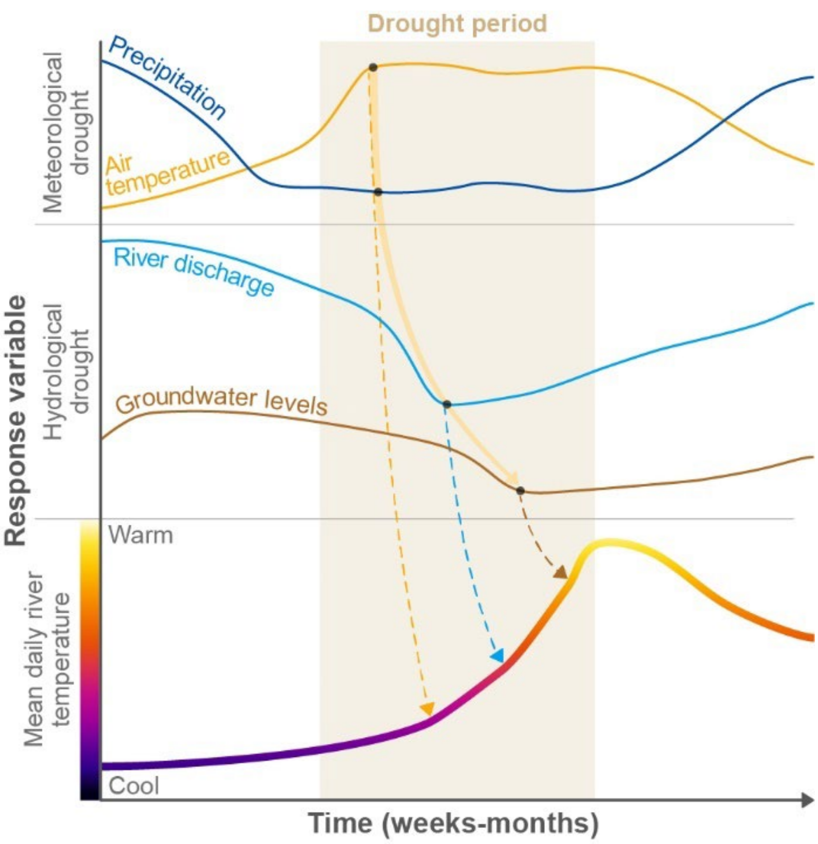


Fig. 1 – A conceptual diagram indicating Tw responses to meteorological and hydrological drought propagation.

In this review, we synthesise evidence documenting drought induced Tw dynamics across the UK and the underpinning mechanisms driving this, which is supplemented with studies from other temperate maritime environments (‘Cfb’ under the Köppen climate classification). We primarily focus on evidence from summer conditions (June-August), where dry and warm meteorological conditions and extreme low-flow conditions are most likely to result in high extreme Tw. While focusing on summer conditions provides greater focus and more reliable comparisons between studies, we acknowledge that droughts during other seasons can have major implications for Tw (Webb and Zhang, 2004), and also recognise that Tw is highly sensitive to intense summer rainfall events (Wilby et al., 2015).

Herein, we conceptualise the mechanisms governing Tw operating across three ‘process sets’. First, we synthesise **energy flux dynamics** as non-advective controls on Tw, and specifically how heat transfer processes in rivers change during drought. Second, we explore the role of **reach-scale habitat** conditions in mediating non-advective controls on Tw, including hydraulic (e.g., residence time) and physical conditions (e.g., riparian vegetation coverages, wetted perimeters). Third, we will examine the influences of water source contributions (surface water and groundwater) as advective heat and water flow controls, on Tw dynamics during drought. We then review how different human influences (pressures and management activities) modify some of the key mechanisms across the three process sets. By integrating and synthesising such evidence, we will characterise the current state of scientific knowledge of Tw during drought (What do we know?) and critical knowledge gaps (What don’t we know?). We later highlight how a better process-based understanding can underpin more robust and accurate models projecting Tw dynamics during future droughts and at unmonitored sites, which can inform management and adaptation efforts (What do we need to know?).

2. Water temperature dynamics during UK droughts

Hot and dry conditions in UK (and countries within similar climates in western Europe) are typically driven by high pressure systems and/or easterly winds from continental climates. Such meteorological settings have instigated various nationally iconic (‘benchmark’) droughts throughout the last century, notably during 1976, 1983-1984, 1988-1992, 2003, 2004-2006, 2010-2012, 2018-2019 and 2022 (Barker et al., 2019; Turner et al., 2021). Wilby et al (2015) noted the longest period of below-average river discharges spanned 5.5 years between 1988 and 1993, although extreme low-flow conditions within droughts often last weeks to months and typically occur during summer months (White et al., 2022). Heatwaves within droughts typically span even shorter timeframes, with a maximum duration of 15-days being reported in the UK during 1976 and 2018 (Beckett and Sanderson, 2022). The occurrence of hot and dry summers has increased drought intensities over recent decades (Barker et al., 2019) and such trends are forecasted to prevail. Specifically, the ‘United Kingdom Climate Projections’ (UKCP18; Lowe et al., 2019) projections indicate increases in average summer air temperatures by 0.9-5.4 °C by

2070 (high emission scenario), with corresponding precipitation levels likely to change between -47 to 2%.

Here, we present different evidence indicating a strong association between Tw values and various benchmark UK droughts, but also highlight notable inconsistencies. Table 1 highlights various studies highlighting drought-induced Tw increases, most notably during the iconic 1976 drought that yielded the longest mean flow deficit on record nationally (Barker et al., 2019). However, in southwest England, Webb and Walling (1993) found that although maximum annual Tw was high during 1976 (20.6 °C) relative to long-term averages, corresponding values were greater during the 1983 drought (22.3 °C) due to higher air temperatures (and associated radiative forcings – see below). Some extreme Tw differences between drought versus non-drought years have been reported (Table 1), although others have reported modest or negligible changes (e.g., Hutchins et al., 2011; Garner et al., 2015).

To support evidence from published literature, in Fig. 2 we present Tw spot-sample observations from select sites with long-term (1975-2022) data within the Surface Water Temperature and Water Quality Data (WIMS) archives (for information on data quality control, see Environment Agency, 2022; river discharge data was downloaded from the nearest flow gauge identified in the 'Hydrology Data Explorer'). For this, all four rivers yielded extreme low-flows during 1976, three of which (all except River Mersey in northwest England) displayed notably high Tw values. The examined rivers generally displayed Tw spikes during other benchmark droughts in 1983, 1989, 1995, 2006 and 2018, although accompanying discharge values did not always fall below the extreme low-flow threshold. However, some Tw inconsistencies were observed, such as during the summer 2003 heatwave that only facilitated notable Tw peaks in one river (River Thames, southeast England), associated with higher air temperatures here relative to other UK regions. Such evidence highlights the increased likelihood of exhibiting Tw spikes during droughts, but inconsistencies reinforce the need to glean a better process-based understanding of the underpinning mechanisms shaping thermal extremes.

Table 1 – Various UK studies documenting water temperature (Tw) in rivers during droughts

Drought event	Location	T_w variable	T_w responses	Reference
1976	Herefordshire, southwest England	Maximum annual T_w	27.6 °C, 4.6 °C higher than non-drought year (1975)	Brooker et al (1977)
	Plynlimon, central Wales.	Mean monthly T_w	~ 14-18 °C, approximately 2-4 °C higher than non-drought years (1977-1978)	Cowx et al (1987)
	Dorset, southwest England	Maximum annual T_w	23.7 °C, 6.2°C higher than non-drought years (1962-1968)	Crisp et al (1982)
	Cumbria, northwest England	Maximum annual T_w	~ 24-29 °C, approximately 4-7 °C higher than non-drought years (1977).	Elliot (2000)
	Aberdeenshire, northeast Scotland	Maximum monthly averaged T_w	~ 15 °C, approximately 1 °C higher than non-drought years (between 1970-2000).	Langan (2001)
	Devon, southwest England	Maximum annual T_w	20.6 °C, 1.4 °C higher than long-term average.	Webb and Walling (1993)
1983-1984	Cumbria, northwest England	Maximum annual T_w	~ 24-29 °C in 1983 across rivers, approximately 4-7 °C higher than non-drought year (1977).	Elliot (2000)
	Devon, southwest England	Maximum annual T_w	20.0-22.3 °C, 0.8-3.1 °C higher than long-term average.	Webb and Walling (1993)
1989-1992	Devon, southwest England	Maximum annual T_w	21.1 °C, 1.9 °C higher than long-term average.	Webb and Walling (1993)
	Aberdeenshire, northeast Scotland	Maximum monthly averaged T_w	~ 15-17 °C, approximately 1-3 °C higher than non-drought years between 1970-2000.	Langan (2001)

	Plynlimon, central Wales	Maximum monthly averaged T_w	~ 12-14 °C in 1991-1992, approximately 2-3 °C higher than non-drought year (1993).	Crisp (1997)
	Plynlimon, central Wales	Summer (mid-morning) maximum T_w	~ 23 °C in 1989, approximately up to 5-11 °C higher than non-drought years (1985-1988; the 1990 drought was ~ 18 °C).	Neal et al (1992)
1995	Aberdeenshire, northeast Scotland	Maximum monthly averaged T_w	~ 14 °C, in keeping with values during non-drought years (between 1970-2000).	Langan (2001)
	Dorset, southwest England	Maximum annual T_w	~ 20 °C, approximately 1.5 °C higher than non-drought years (1993-1994 and 1995)	Bowes et al (2011)
2003, 2004-2006	Dumfries and Galloway, southwest Scotland	Maximum monthly averaged T_w	~ 18 °C during 2003 open sites, approximately 2 °C higher than non-drought years (2000-2002). No discernible T_w differences between years in shaded reaches.	Webb and Crisp (2006)
	Aberdeenshire, northeast Scotland	Monthly averaged T_w	~ 12-15 °C during 2003-2005, no discernible differences with non-drought years (2007-2009), although 2006 was approximately 2-4 °C warmer than long-term averages.	Garner et al (2015)
	Hampshire, southern England	Maximum monthly T_w	~ 20-22 °C in 2006, >2-3 °C higher than other years examined (2005 – drought) and (2007 – non-drought).	Broadmeadow et al. (2011)
2010-2011	Oxfordshire, Central England	Maximum annual T_w	~ 20-22 °C, no discernible differences with non-drought years (2009 and 2012).	Hutchins et al (2011)
2018	Aberdeenshire, northeast Scotland	Maximum annual T_w	~17 °C, approximately 3 °C higher than non-drought year (2019).	Fennell et al (2020)

	East Anglia, eastern England	Maximum annual T_w	~18 °C, no discernible differences with non-drought years (between 2012-2017).	Cooper et al (2020)

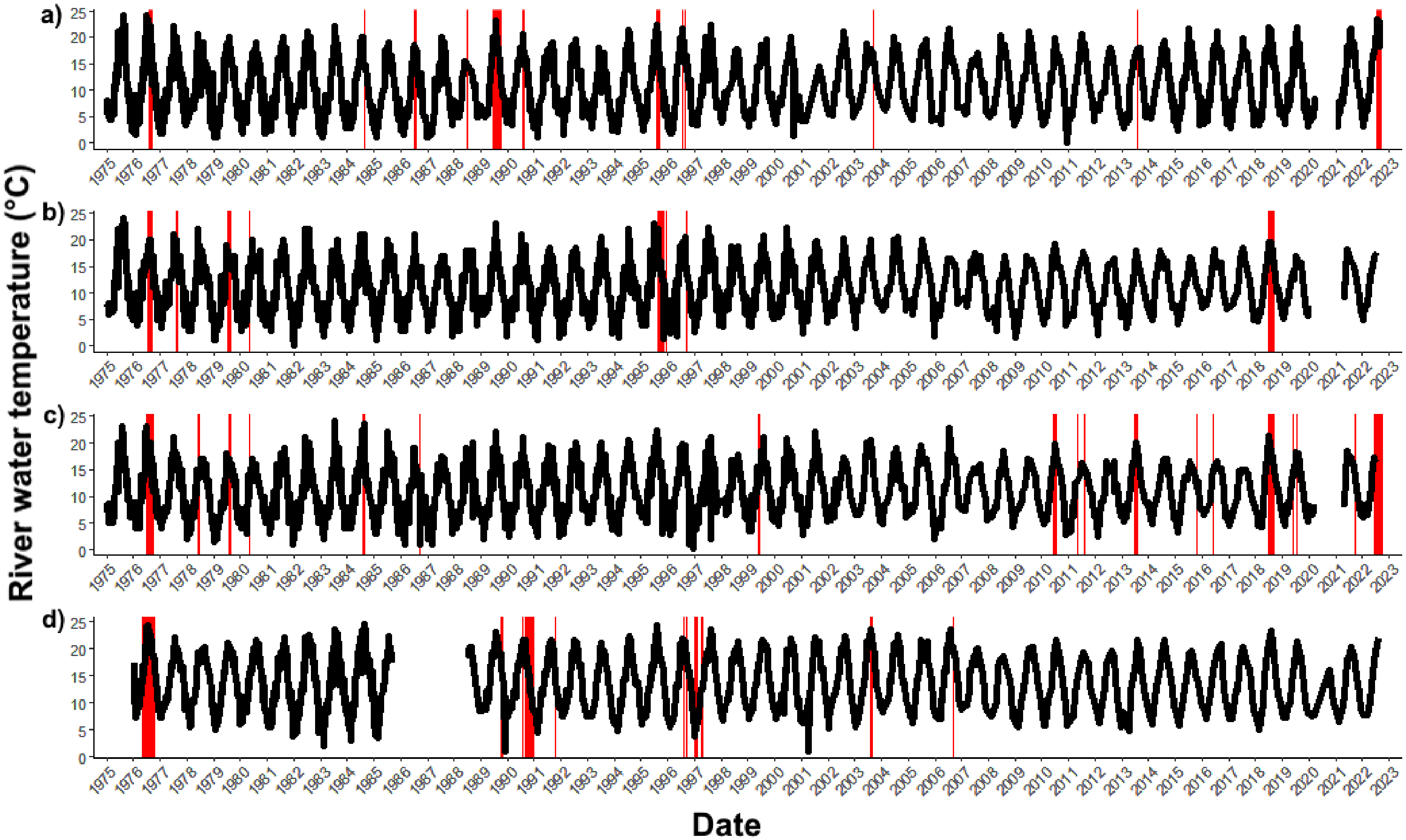


Fig. 2 – Long-term monthly averaged river water temperature (Tw) variations from spot samples across four lowland rivers. Extreme low-flow events, classified as discharge values below that exceeded 99% of the time (Q99), were identified based on nearby flow gauge data (Environment Agency, 2023). a) River Ouse, Yorkshire (northeast England; gauge ID = 'F2405'); b) River Mersey, Greater Manchester (northwest England; gauge ID = '692524'); c) River Irwell, Greater Manchester (northwest England; gauge ID = '690511'; and d) River Thames, London (southeast England, gauge ID = '3400TH').

Drought induced Tw extremes have had wide ranging ecological consequences in the UK. This includes increased endemic diseases amongst fish communities, reduced fish survival, feeding and growth (Elliot and Elliot, 2010), enhanced algal blooms (Bowes et al., 2011), restricted gross primary production and ecosystem respiration (Arias Font et al., 2021) and reduced biodiversity and foodweb complexity (Lu et al., 2016).

3. Controls and processes governing drought-induced river water temperature: a conceptual framework

3.1. Overview of different controls shaping drought-induced Tw

In the following three subsections, we synthesise literature examining the primary drivers of Tw dynamics during droughts corresponding to the energy flux dynamics, **reach-scale habitat** and **water source contributions** (advective flows and heat) process sets. We layer human influences on these ‘natural’ drivers in Section 4. Table 2 summarises critical findings from key publications examining the mechanisms governing Tw dynamics during drought and non-drought summer periods in the UK (the latter still providing valuable insights during low-flow and warm meteorological conditions). Fig. 3 summarises the research focus of studies outlined in Table 2 in terms of the process set examined, and indicates spatial biases in Tw research due to institutional specialisms, with energy budget studies focussing on southwest England or northeast Scotland. Moreover, many of such studies examining the energy flux influences drought-induced river temperature dynamics have largely stemmed from studies with broader focuses (e.g., annual thermal dynamics). Tw comparisons between riparian coverages often have often occurred in central Wales. There is a dearth of studies in Northern Ireland, northern Scotland, northern and Eastern England. Moreover, Tw studies examining compound drought-heatwave studies are more limited still (but see Garner et al., 2014; 2017), and such events have strong ramifications for high river thermal extremes.

Table 2 – A summary of UK studies examining water temperatures (Tw) in rivers during summer drought or seasonally typical low-flow conditions. References are organised firstly by drought versus non-drought studies, secondly by process sets and then by the alphabetical order of the author's surname.

[a] Nationally iconic, 'benchmark' drought years identified from (Barker et al., 2019; Turner et al., 2021; UK's Centre for Ecology & Hydrology, 2022). [b] GW = Groundwater and SW = Surface water

	Contextual information					T_w controls		T_w responses			
	Low-flow period[a]	**Catchment position**	**SW/GW influences[b]**	**Dam influences**	**Riparian shading**	**Mechanism examined**	**Key properties**	**Variable**	**Effect**	**Response**	**Reference**
	Drought studies										
Energy flux dynamics	Drought conditions experimentally mimicked	Headwaters	GW	Free flowing	Low (potential shading from flume walls)	Solar radiation	Daily averages on a clear day were 2.4-5.5 × higher than overcast days.	Percentage contribution to total heat budget	Warming	Contributed 64% to T_w variations in daytime hours	Folegot et al (2018)
Energy flux dynamics	Compound drought-heatwave period monitored (July 2013)	Headwaters	SW	Free flowing	Mixed	Various energy fluxes	Shortwave radiation dominated heat inputs and were ~ 3× higher than an overcast day.	Maximum longitudinal T_w difference (1.5km reach)	Warming	2.5 °C on days with clear skies, but 0.6°C on overcast day.	Garner et al (2014)
Energy flux dynamics	Drought year (1992)	Headwaters and lowlands	Mixed	Free flowing and regulated	Mixed	Various energy fluxes	Solar radiation dominated total energy inputs, followed by sensible heat	N/A	N/A	N/A	Webb and Zhang (1997)

							(mean contributions = 68.7% and 21.2%, respectively).				
	Drought year (1995)	Headwaters and lowlands	Mixed	Free flowing and regulated	Mixed	Various energy fluxes	Shortwave radiation dominated heat inputs, followed by sensible heat (mean contributions = 48.4-84.9% and 5.1-40.8%, respectively).	Monthly mean T_w	Warming	Positive effects, but this was more variable in shaded and regulated reaches.	Webb and Zhang (2004)
	Drought (2005-2006) and non-drought (2007) years	Headwaters	SW	Free flowing	Mixed	Riparian shading	Reaches spanning covered to open riparian influences were compared.	Maximum summer T_w	Cooling	Increases during drought were lower in shaded (0.1-1.7 °C) versus open (0.8-3.4 °C) reaches	Broadmeadow et al (2011)
	Drought (1995) and non-drought years (1991 and 1993).	Headwaters	SW	Free flowing	Mixed	Riparian shading	Reaches with cleared and intact riparian coverages were compared.	Maximum summer T_w	Cooling	~ 1 °C lower in shaded versus cleared reaches across both drought and non-drought years	Crisp (1997)

	Drought conditions mimicked	Headwaters	GW	Free flowing	Low (potential shading from flume walls)	Flow depth	Spanned 7-25cm reflecting drought to non-drought hydrological gradients.	Maximum longitudinal T_w difference (15m long flumes)	Warming	>3 °C in drought treatments, which were ~ 2 °C higher than non-drought treatments.	Folegot et al (2018)
Reach-scale habitats	Drought (2003-2006) and non-drought years (2007-2009)	Headwaters	Mixed	Free flowing	Mixed	Various energy fluxes; Riparian shading	Shortwave radiation dominated heat inputs, but were almost 3 × lower in shaded versus open reaches.	Monthly mean T_w	Cooling	1.1 °C lower in shaded versus open reaches, the former was typically cooler.	Garner et al (2015)
	Drought (1984, 1989-1990) and non-drought years (1985-1988)	Headwaters	Mixed	Free flowing	Mixed	Riparian shading	Reaches with cleared and intact riparian coverages were compared.	Maximum summer T_w	Cooling	~ 4-9 °C lower in shaded versus cleared reaches, and differences were most pronounced during drought years	Neal et al (1992)
	Drought (1995) and non-drought year (1996).	Headwaters	SW	Free flowing	Mixed	Riparian shading	Riparian coverages were cleared between the two summers.	Summer (July-August) monthly maximum T_w	Cooling	5.3-7.0 °C cooler pre (shaded) versus post (open) clearance, despite the former occurring	Stott and Marks (2000)

										during a drought.	
	Drought (2003) and non-drought years (2000-2002)	Headwaters	SW	Free flowing	Mixed	Riparian shading	Reaches spanning covered to open riparian influences were compared.	Summer monthly mean maximum T_w	Cooling	~5 °C lower in shaded versus open reaches, and differences were higher during drought versus non-drought years (~3-4 and 5 °C, respectively).	Webb and Crisp (2006)
Water source contributions	Drought year (2018)	Headwaters	GW	Free flowing and regulated	Unclear	Deep (≥1.5m) groundwater contributions	Sustained 65-100% of river discharges	Range of summer T_w values	Cooling	Inputs remained cool (~ 6-7°c) throughout the drought.	Fennell et al (2020)
	Drought year (1992)	Headwaters and lowlands	Mixed	Free flowing and regulated	Mixed	Groundwater contributions	Groundwater exerted negative and positive contributions to the total energy budget (spanning -13.8 to 12.8%).	N/A	N/A	N/A	Webb and Zhang (1997)
	Non-drought studies										
Energy	Non-drought year (1994)	Headwaters	Mixed	Regulated	Low	Various energy fluxes	Shortwave radiation dominated heat inputs	Summer T_w (15-min recordings)	Warming	Positive effects that were strongest on	Evans et al (1998)

							(97.2-98.8%), almost 2× higher than an overcast day.			the hottest summer day, explaining 86% (r = 0.93) of T_w variations.	
	Study period spanned 1984-2007, but droughts not assessed directly.	Headwaters and lowlands	Mixed	Unclear	Mixed	Solar radiation	Shortwave radiative inputs yielded positive effects on T_w (but air temperature effects were much higher).	Summer averaged T_w	Warming	Shortwave radiation effects were consistently low across T_w values spanning 15-20 °C.	Laize et al (2017)
	Non-drought year (1994)	Headwaters	GW	Free flowing	Low	Various energy fluxes	Shortwave radiation dominated heat inputs.	N/A	N/A	N/A	Webb and Zhang (1999)
Reach-scale habitat	Non-drought year (2010)	Headwaters	GW	Free flowing	Mixed	Various energy fluxes; Riparian shading	Shortwave radiation dominated heat inputs (96.2-97.5%), and were ~ 2-3× lower in shaded versus open sites.	Summer maximum T_w	Cooling	2 °C cooler in shaded versus open reaches.	Dugdale et al (2018)
	Non-drought year (2004)	Headwaters	Mixed	Free flowing	Mixed	Various energy fluxes; Riparian shading	Shortwave radiation dominated heat inputs (95.9-100%), and were almost 5× lower in	Summer (August) maximum T_w	Cooling	0.5 °C cooler in shaded versus open reaches.	Hannah et al (2008)

							shaded versus open sites.				
	Non-drought year (2008; ‘hottest week’)	Headwaters and lowlands	Mixed	Free flowing	Mixed	Riparian shading	Modelled reforestation that covered 100% of the river network	Mean weekly maximum T_w	Cooling	1.1 °C cooler compared to measured land cover	Hrachowitz et al (2010)
	Non-drought year (2015)	Headwaters and lowlands	Mixed	Free flowing (largely)	Mixed	Channel width	Spatial models incorporated widths from 1 to over 100m	Maximum summer T_w	Warming	Almost linear positive relationship modelled	Jackson et al (2017)
	Non-drought year (2015)	Headwaters and lowlands	Mixed	Free flowing and regulated	Mixed	Riparian shading	Spatial models simulating 0-100% riparian coverages	Maximum summer T_w	Cooling	Up to 2.8 °C, cooling effects were highest when air temperatures peaked.	Jackson et al (2018)
	Non-drought years (2011-2014)	Headwaters and lowlands	GW	Free flowing	Mixed	Solar radiation; Riparian shading	Statistical models simulating planting initiatives identified incoming solar radiation could be reduced by 30-40%.	Maximum summer T_w	Cooling	1 °C reductions with ~ 0.5 (headwater) and 1.1km (lowland) of complete shade.	Johnson and Wilby (2015)
	Non-drought year (1986)	Headwaters	SW	Free flowing	Mixed	Riparian shading	Reaches spanning covered to open riparian influences were compared	Maximum summer T_w	Colling	~ 0.5-2 °C reductions in forested versus open sites	Weatherly and Ormerod (1990)

Water source contributions	Non-drought year (2008; 'hottest week' typical for summer)	Headwaters and lowlands	Mixed	Free flowing	Mixed	Tributary inputs	Modelled catchment-wide, spatially continuous T_w measurements	Mean weekly maximum T_w	Neutral	Mainstem (19-20 °C) was buffered from warm upland (23.7 °C) and cool (15.8 °C) coastal tributary inputs.	Hrachowitz et al (2010)
	Non-drought year (2015)	Headwaters and lowlands	Mixed	Free flowing and regulated	Mixed	Upstream advective inputs	Spatial models incorporating 'River Network Smoother' indicating spatial autocorrelation	Maximum summer T_w	Variable	Warmer and cooler influences in the headwaters and mid-sections of the catchment, respectively	Jackson et al (2017; see also Jackson et al., 2018)
	Non-drought years (2011-2014)	Headwaters and lowlands	GW	Free flowing	Mixed	Upstream advective inputs	Spatial autocorrelation tested between T_w loggers	Daily maximum T_w	Variable	Positive relationship, with correlations (r) spanning 0.90-0.98.	Johnson et al (2014; Johnson and Wilby 2015)
	Non-drought years (2011-2014)	Headwaters and lowlands	GW	Free flowing	Mixed	Groundwater contributions	Spring input inventory created across watercourses	Annual maximum T_w (likely during summer)	Cooling	Largely spanned 10 – 11.5 °C, while spring inputs were predominantly between 8.5-10 °C.	Johnson et al (2014)

	Non-drought year (1994)	Headwaters	GW	Free flowing	Low	Groundwater contributions	Groundwater contributions to the average daily heat storage spanned 5.8-12.1%.	Summer T_w (15-min recordings)	Cooling	Cool groundwater inputs dampened radiative effects, 0.30-0.34 °C increases for every 1°C rise in air temperature.	Webb and Zhang (1999)

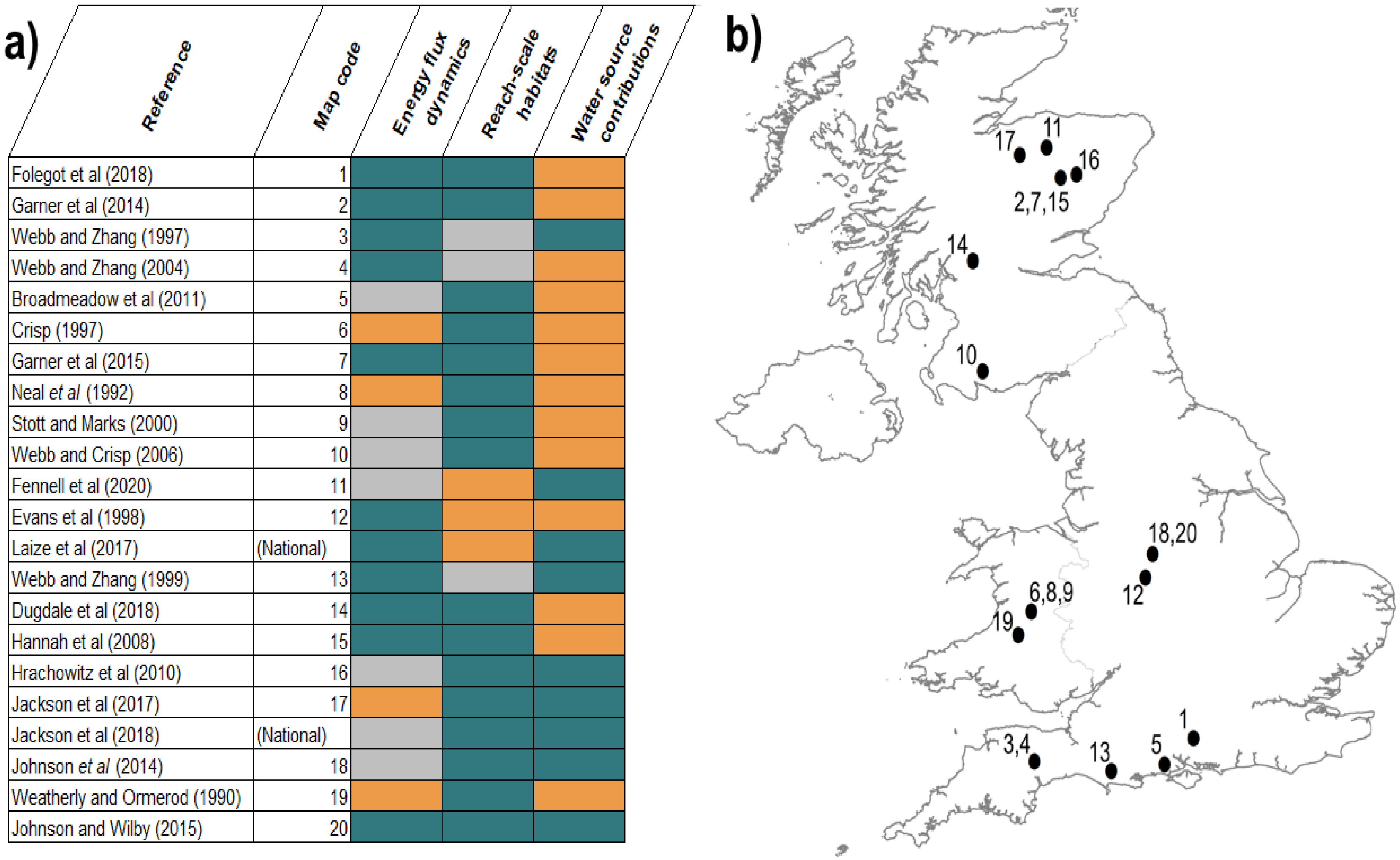


Fig. 3 – A summary of key literature examining mechanisms underpinning drought-induced Tw dynamics. a) A heatmap indicating which process sets were examined in the study (orange = not tested; grey = considered but not directly quantified; blue = empirically examined); b) A map displaying the distribution of studies. References are ordered in the order they appear within Table 2.

3.2. The influences of energy flux dynamics on river temperature during drought

Although positive associations have been demonstrated between thermal energy inputs and Tw (e.g., Webb and Zhang, 2004; Garner et al., 2014), this has not widely explored during drought periods. Various UK thermal energy budget studies have presented monthly summaries (e.g., Web and Zhang, 1997; 2004; Garner et al., 2015), whereby non-advective influences during droughts (or extreme low-flows) and compound drought-heatwaves operating at shorter temporal scales will be overlooked. Notwithstanding, different thermal energy budget studies during undertaken during drought and non-drought summer conditions have provided critical insights into key positive and negative heat influences. Energy inputs elevating Tw in river environments include incident shortwave (solar) and downward longwave (atmospheric) radiation, condensation, and in-channel friction. Predominant summer energy losses cooling rivers include longwave radiation emissions and evaporation (latent heat) effects, while sensible heat (i.e., conduction at the air-water interface and convection) and water column-riverbed exchanges can have warming or cooling influences (Webb and Zhang, 1997; Hannah and Garner, 2015).

Thermal energy budget studies spanning different river typologies have consistently identified shortwave radiation as the primary heat input, often contributing >90% of the warming effects during summer and specifically drought conditions (see Table 2) and typically outweigh any non-advective cooling influences. This has been recognised across river environments worldwide (Leach et al., 2023), but can become less influential in smaller streams most sensitive to shading by riparian cover (e.g., Kaandorp et al., 2019) or valley sides in incised systems. Shortwave radiation inputs typically peak during hot and dry conditions associated with droughts, although this has not been widely verified in UK thermal energy budget studies. In fact, in southwest England, Webb and Zhang (2004) reported that shortwave radiation inputs along shaded reaches declined between late spring-early summer and mid-late summer (48-83% and 26-43% contributions to energy flux inputs, respectively), despite air temperature peaking within the latter when a compound drought-heatwave occurred. Similar temporal trajectories were reported by Garner et al (2015) in northeast Scotland, a region with drastically differing hydroclimatic conditions than southwest England (typically colder at higher latitudes). Such findings potentially reflect greater riparian canopy cover later in summer as bankside deciduous vegetation matures.

Sensible heat inputs are contingent upon air temperature exceeding Tw as warmer air can be conducted into the water column. Thermal energy budget drought studies from southwest England have reported high sensible heat contributions, contributing up to 70% (Webb and Zhang, 1997) and 41% (Webb and Zhang, 2004) of the non-advective heat inputs during droughts (although was most commonly a secondary influence behind radiative inputs). It is unclear why such high values arose and, but its proximity to the southwest coast exposed to warm trade winds potentially indicates that hot air was transported from the continent (rather than elevated temperatures from radiative forcings). Kaandorp et al (2019) also reported high sensible heat contributions in a lowland system in The Netherlands and attributed this to riparian shading lessening solar radiative inputs, which in turn increased air temperature-Tw differences and thus encouraged sensible heat transfer. However, sensible heat inputs most commonly yield minimal influences on Tw, which has been highlighted in various studies spanning different river environments (e.g., northwest Scotland – Garner et al., 2014; 2015; central England - Evans et al., 1998; Luxembourg – Westhoff et al., 2007; France – Wawrzyniak et al., 2017).

Longwave radiation typically peaks during hot and dry conditions as shortwave inputs are absorbed and emitted by the earth's surface and atmosphere, but are not completely congruent as the former responds positively to cloud cover (Laize et al., 2017). Incised and shaded streams can facilitate longwave radiation warming effects during hot and dry conditions, particularly at night, whereby such energy is retained and re-emitted back towards the water surface (e.g., Kaandorp et al., 2019). However, longwave radiative fluxes most commonly exert a Tw cooling effect during the summer due to emissions from the channel (Hannah et al., 2008; Laize et al., 2017), which is enhanced by warmer waters (as governed by the Stefan-Boltzmann law - Hannah et al., 2004). Drought studies from northeast Scotland (Garner et al., 2014; 2015) highlighted that longwave radiation emissions yielded minimal cooling influences across shaded and non-shaded reaches, while those from southwest England indicate more significant influences. For instance, Webb and Zhang (2004) reported that longwave radiation consistently contributed the highest proportion of non-advective heat losses (28-66%) across various river typologies, while corresponding inputs were markedly lower (1-32%). They reported longwave emission cooling effects were greatest in open reaches during periods when air temperature peaked, and Evans et al (1998) reported similar trends during a non-drought summer. Such contrasts between UK studies likely reflect Tw differences, with warmer waters occurring at lower latitudes facilitating greater longwave emissions.

Latent heat exchanges during hot and dry conditions largely comprise evaporative cooling effects, with condensation only yielding minor warming influences (Webb and Zhang, 1999; Dugdale et al., 2018). Drought studies from southwest England indicate that evaporative cooling influences are less influential than longwave emissions (see also Evans et al., 1998). Webb and Zhang (1997) reported that evaporation contributed between 19-48% of the energy flux cooling effects, with greater influences occurring at the beginning or end of summer period in exposed headwater streams. In the same catchment, Webb and Zhang (2004) highlighted slightly lessened evaporative effects on Tw (10-29% contributions energy flux cooling influences), which was limited by lower wind speeds. Drought studies from northeast Scotland reported even higher evaporating cooling effects during droughts, with Garner et al (2014) highlighted that evaporative cooling effects surpassed solar radiative inputs in one shaded reach (but comprised approximately one-third of the heat inputs in other shaded and non-shaded reaches). Garner et al (2015) reported monthly average heat energy losses were consistently over half that of shortwave radiative heat input values, while other studies regionally have reported high evaporative cooling effects during non-drought summers (Hannah et al., 2008; Dugdale et al., 2018). Evaporative cooling effects will be higher in rivers susceptible to low atmospheric humidity and high wind speeds (e.g., watercourses devoid of riparian zones and hence no obstructions reducing wind speeds - Hannah et al., 2004). Critically, while it is widely recognised that evaporative cooling effects can offset thermal extremes during hot and dry conditions, how such influences respond to varying drought severities is poorly understood as most energy budget studies use empirical or semi-empirical approaches to estimate latent heat fluxes (Dugdale et al., 2019).

Bed conduction effects on Tw are driven by thermal differences between the riverbed and the overlying water column (Evans et al., 1998). This can vary depending on vertical thermal gradients in the water column, flow depth and turbidity governing the proportion of solar radiation reaching the riverbed, and the absorbance / reflective properties of the benthic habitat (e.g., substrates, primary producers; Evans et al., 1998). While declining flow depths during droughts could increase the radiative forcings reaching the riverbed, the water column typically warms at faster rates and therefore bed conduction yields cooling effects overall (e.g., Evans et al., 1998; Westhoff et al., 2007; Kaandorp et al., 2019). For instance, Webb and Zhang (1997) and (2004) found that bed conduction respectively contributed 11-35% and 14-29% of the total heat energy losses, while corresponding heat inputs were minimal. Cooling effects from bed conduction are enhanced by groundwater inputs that often yield lower temperature values compared to Tw (see below) and therefore cool benthic habitats, such as Webb and Zhang (1999) who reported that bed conduction contributed 52% of the total non-advective heat losses in a groundwater-dominated river during a non-drought summer.

All UK thermal energy budget studies have reported the minimal role of friction from the bed and banks in shaping Tw regimes (see references above and Table 1). The only exception to this is Webb and Zhang (2004), who did report such influences yielding higher energy inputs in two of their four studied rivers during summer. The most notable of these was a regulated system with artificially elevated low-flow discharges. However, overall, in-channel friction will be lessened during drought conditions when flow velocities and turbulence is reduced (Webb and Zhang, 2004).

3.3. Reach-scale habitat influences on river temperature during drought

In this sub-section, we focus on the three most widely researched reach-scale habitat conditions mediating non-advective influences on Tw: (i) riparian vegetation shading; (ii) water volume heat capacity effects; (iii) hydraulic conditions governing residence times. We recognise that other habitat conditions operating at smaller spatial scales (e.g., individual habitats - macrophyte assemblages – Webb and Zhang, 1999; thermal conductivity of bed substrates – Evans et al., 1998) are also likely to affect drought-induced Tw dynamics, but their effects are not considered to be as important as those discussed here.

3.3.1. Riparian vegetation influences on drought-induced river temperature dynamics

Riparian vegetation shading effects on Tw is dependent on complex interactions between channel width, gradient, orientation, aspect, tree height, vegetation density and functional properties and solar geometry (Garner et al., 2017; Jackson et al., 2021). Consequently, extreme low-flow conditions associated with droughts influence shading effects by controlling the wetted perimeter (and hence the proportion of the water shaded) and riparian vegetation health. While the former has not been widely explored specifically in the context of drought, the effects of bankfull channel widths on governing the proportion of water shaded by riparian vegetation has been quantified in various modelling studies (Bachiller-Jareno et al., 2019; Jackson et al., 2021). Upland environments (with typically narrower channels) and lowland systems with reduced width:depth ratios channels and steeper banks / margins (e.g., clay-based systems - Sear et al., 1999) will facilitate greater shading influences; 'triangular' hydraulic geometries in river environments with non-cohesive banks could be less sensitive to riparian shading as greater water volumes are concentrated in mid-channel regions (Ferguson, 1986). The effects of drought on riparian vegetation have received limited research, despite such extremes been widely demonstrated to cause wilting, stunted leaf growth or dieback (Ilyas et al., 2017). Moreover, wildfires are becoming more common due to climatic changes adds further pressure on riparian vegetation. While such ecological threats during droughts are unlikely to completely irradicate the shading effects of riparian vegetation (e.g., Webb and Zhang, 1999 highlighted the shading effects of bare tree trunks in winter), reduced canopy density could increase solar radiation levels reaching the water surface.

Riparian shading denotes a widely recognised means of reducing high Tw extremes (see Table 2). For instance, Wilby and Johnson (2020) utilised an inventory of nearly 1 million spot sample Tw measurements across England and quantified the average cooling effects of riparian shading was as high as ~2.8 °C. Such Tw reductions are largely driven by riparian vegetation inputs constraining shortwave radiative inputs. Webb and Zhang (2004) highlighted that shortwave radiative inputs were ~3-6 times higher in exposed sites relative to other systems during the 1995 drought, while Hannah et al (2008) and Dugdale et al (2018) reported that shaded sites exhibited shortwave radiative inputs ~4 and 1-3 times lower than open reaches in non-drought summers (Tw being 0.8°c and 0.5-1°c cooler), respectively during non-drought summer conditions. While riparian vegetation can facilitate some warming effects, including reflecting longwave radiation re-emitted back towards the channel (Dugdale et al., 2018) or reducing windspeeds and hence evaporative cooling rates (Hannah et al., 2008), such effects are typically outweighed by reduced shortwave radiative inputs.

3.3.2. Water volume heat capacity influences on drought-induced river temperature dynamics

River discharges significantly alter drought-induced dynamics, not only through the effects of shifting upstream advective inputs (see below), but through its effects on the thermal capacity of the watercourse. For instance, in southwest England Webb et al (2003) reported that air-water relationships (the former as a surrogate for radiative effects) steepened when discharges were below median levels, while Booker and Whitehead (2022) found that in New Zealand declines in river flow from the median to the fifth percentile facilitated an average Tw increased by 0.5°C (after accounting for seasonality and meteorological variability; see also Van Vliet et al., 2011). Alternatively, Foleget et al (2018) experimentally simulated drought conditions using outdoor flow-through flumes and reported those containing severely depleted water levels exhibited a maximum warming of

3.3°c, ~2 °C higher than those conveying higher discharges. Water volume controls on the thermal capacity of rivers are strongly governed by channel depth and turbidity, which can alter Tw by influencing the extent to which radiative forcings can penetrate the water column. Deeper water parcels are typically more protected from radiative forcings that are scattered higher in the water column (Evans et al., 1998). Consequently, the hydraulic geometry of river channels and how this governs width:depth ratios (and velocities – see below) to extreme low-flow conditions is fundamental to drought-induced Tw dynamics (Garner et al., 2017). Wider channels can promote such conditions as they facilitate reduced channel depths, and hence rivers with greater width:depth ratios can become more exposed to greater non-advective influences. Although many groundwater dominated systems in the UK conform the archetype of rivers with high width:depth ratios (namely ‘chalk’ systems – Sear et al., 1999), they are generally less susceptible to warming due to the dominant advection of groundwaters (see below). Moreover, channels exhibiting hydraulic geometries facilitating more rapid depth reductions during extreme low-flow conditions, including systems with straightened (e.g., cohesive or vegetated alluvial) banks or outer sections of meander bends (Ferguson, 1986).

3.3.3. Flow velocity influences on drought-induced river temperature dynamics

Stream velocities and geomorphology (notably slope influences on stream power) controls water residence times, which strongly governs Tw as this dictates the amount of time water parcels are exposed to their surroundings, and thus the accumulation and dissipation of heat. Garner et al (2017) modelled the effects of differing flow velocities on Tw dynamics in a forested reach in northeast Scotland during a compound-drought heatwave and reported slow-flow conditions through a shaded system reduced maximum Tw values by 4.9°c. In the eastern region of The Netherlands, Kaandorp et al (2019) reported that enhanced residence times of a larger, instream pond elevated maximum Tw summer values of 6.4 °C relative to its smaller counterpart. Hence, river systems yielding greater residence times, including those occurring along shallow gradients (e.g., lowland environments, plateaus) or notably wide or deep systems (e.g., online ponds, unconfined reaches), will become more susceptible warming during drought by being exposed to atmospheric forcings for longer periods. It should be noted that stream velocity will increase channel friction that can elevate Tw values, although such influences are typically minimal (particularly during low-flow conditions – see above).

3.4. Water source contributions controls on river temperature during drought

The effects of upstream advective influences (surface water contributions) on Tw is dependent on the thermal properties and relative discharges of the mainstem channel and inflowing tributaries. For instance, Hrachowitz et al (2010) reported that the mainstem river facilitated stable temperatures between 19-20°c during the hottest week in a meteorologically typical summer, which was buffered from contributions from warm upland (15.8°c) and cool (23.7°c) coastal tributaries. Conversely, Johnson et al (2014) highlighted that Tw displayed highest spatial variation along a karstic watercourse with greater tributary influences during a non-drought summer. Reduced advective upstream inputs becomes most prevalent when flow cessation events occur, such as systems exclusively fed by groundwater inputs (i.e., no upstream contributions – White et al., 2018) or hydrologically disconnected instream pools that undergo rapid warming (Datry, 2017). Further research quantifying the effects of such surface water disconnections on Tw is required as these effects are poorly understood.

Various studies have reported that surface water contributions declined at faster rates than groundwater inputs during low-flow conditions (e.g., Webb and Walling, 1997; Dewson et al., 2007; Wawrzyniak et al., 2017), and therefore the volume and thermal properties of groundwater inflows (or lack thereof) have significant implications for drought-induced Tw. Groundwater inputs have been widely reported to yield a cooling effect on Tw during hot and dry conditions in the UK (Webb and Zhang, 1999; Johnson et al., 2014) and internationally (Dewson et al., 2007). For instance, in a lowland system in southeast France, Wawrzyniak et al (2017) reported that groundwater cooling effects were most influential (average reduction of 0.68 °C) during a drier and warmer summer when such contributions comprised a higher proportion of the river discharge. Shallow groundwaters are often cooler than average summer air temperatures, and in the UK groundwaters typically cool at greater depths until ~ 15 m and warm thereafter (Busby et al., 2009). For instance, in a limestone-fed headwater system in northeast Scotland, Fennell et al (2020) reported that while Tw was highest in late July (~15°c), shallower groundwaters (≤1.2m deep) displayed a lagged response and peaked in September (~10°c), and the temperature of deeper groundwaters (≥1.5m deep) remained cool (~ 6-7°c) and stable throughout. They highlighted that deeper (and older) groundwater inputs sustained 65–100% of river discharges during a drought, and therefore buffered Tw from meteorological forcings during the 2018 drought. However, how groundwater temperature varies in relation to Tw is dependent on various factors, including subsurface depths and flow rates, as well as the thermal conductivity of the lithology. Wood et al (2005) reported that water temperature in one karstic spring was >5°c higher than air temperatures due to geothermal inputs, which occur commonly across such Carboniferous limestone regions (e.g., Johnson et al., 2014; see also Sear et al., 1999). In a region with variable underlying groundwater contributions (southwest England), Webb and Zhang (1997) reported that both warming and cooling influences on Tw, the former likely reflecting shallow groundwater inputs affected by atmospheric conditions, or influences from deeper groundwaters that are exposed to higher heat flow rates (i.e. heat conducted from the Earth's core; Busby et al., 2009). Overall, there have been a lack of studies quantifying groundwater influences on Tw during drought conditions (but see Fennell et al., 2018), and even less evidence has quantified the thermal implications of groundwater disconnections.

3.5. Re-conceptualising mechanisms governing drought-induced river temperature dynamics

Based on the evidence synthesised above, we have conceptually detailed the dominant mechanisms that are likely to influence Tw during different drought severities in Fig. 4, which will apply to various river environments across Cfb climates and globally. Shortwave radiative inputs are likely to dominate in various river system typologies across varying drought severities, while sensible heat inputs will often typically exert a secondary or minimal warming influence on Tw (but will increase with air temperature). Cooling effects from longwave radiation emissions and evaporation are both likely to increase during droughts, both driven by elevated Tw values and the latter also being influenced by lower atmospheric humidity. Bed conduction will also yield more influential cooling effects during drought conditions as benthic habitats will typically be possess lower temperatures than the overlaying water column (particularly in groundwater-dominated systems), although will typically not be as influential as longwave emissions and evaporative influences. As drought intensifies the influences of upstream advective influences (surface water contributions) on Tw will be dampened, which could have varying effects depending on the thermal properties of contributing tributaries. Groundwater inputs may become equally or more important during low-moderate drought intensities (as surface waters recede at faster rates), while extreme hot and dry conditions drastically lowering water tables can significantly constrain the effects of such subsurface inputs. When dense and tall riparian vegetation is present, this is likely to be highly influential across various drought intensities. Declining water levels will increase Tw as droughts become more severe, while reduced stream velocities will exacerbate this warming effect by increasing residence times. We should highlight that although these processes are described independently, various interdependencies between such mechanisms will govern Tw drought-induced dynamics. Such interactive effects have been most widely recognised for the effects of riparian shading influences and low-flow discharges on solar radiative forcings, but have been significantly under researched across other mechanisms spanning different process sets. For instance, it is unclear how hydraulic geometry responses to varying drought severities mediate the thermal buffering capacity of different river environments via changes in channel widths, stream velocities (and hence residence times) and turbidity levels. The influence of other physical habitat conditions (e.g., woody material, macrophyte communities, different substrate compositions) on both thermal energy fluxes and water source contributions has also been sparsely researched, but could significantly affect Tw dynamics during droughts by modifying instream hydraulics and hyporheic exchanges (Magliozzi et al., 2019). Lastly, while it is known shallow groundwaters are susceptible to meteorological forcings during hot conditions (Busby et al., 2009), it is poorly understood how this varies across different drought gradients and between different lithological properties. Gleaning a more comprehensive process-based understanding of such interactive influences could facilitate more robust and accurate Tw models capable of guiding management interventions.

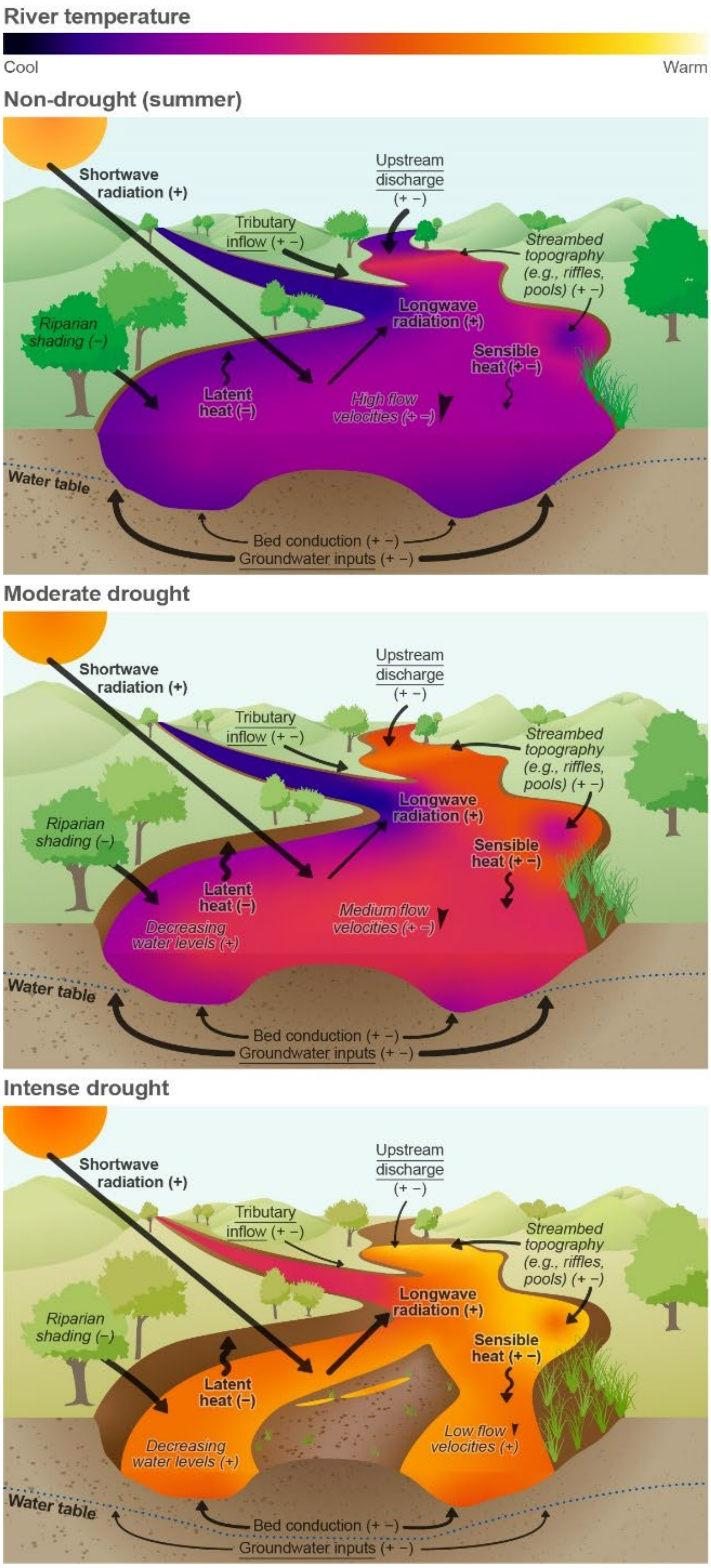


Fig. 4 – The relative influence of dominant heat fluxes governing drought-induced river water temperature (Tw) variations. Thin, moderate and thick lines denote small, intermediate and large effects, respectively. Emboldened, italicized and underlined text denotes non-advective, reach-scale habitat and advective influences.

4. Human influences affecting river temperature during droughts

The following section focusses on four key human influences affecting Tw during droughts: (i) riparian vegetation modifications; (ii) flow regulation; (iii) water abstraction; and (iv) channelisation (physical modifications to river channels). While often reported separately, there are undoubtedly interdependencies between these human activities (e.g., urban, channelised rivers are usually devoid of riparian vegetation). The following sub-sections detail the pressures associated with human activities, as well as potential management interventions that can be introduced to mitigate or reverse such impacts.

4.1. Riparian vegetation clearance and planting

Human activities affecting riparian vegetation coverages is arguably the most widely researched human influence affecting Tw (Garner and Hannah, 2015; Orr et al., 2015). Various studies have employed space-time substitutions comparing covered (forested) versus open reaches to quantify the extent to which riparian vegetation can reduce summer maximum Tw values (Broadmeadow et al., 2011; Imholt et al., 2013; for a detailed review on this topic in the UK, see Webb and Crisp, 2006), and solar radiation reaching the water surface (Hannah et al., 2008; Dugdale et al., 2018; Bachiller-Jareno et al., 2019). Various studies have employed modelling techniques to predict where riparian planting management initiatives could decrease Tw values. For instance, Garner et al. (2017) suggested that planting on the southerly bank of river reaches with an E-W orientation that possess lower flow velocities (i.e., longer residence times) would yield the greatest reduction in peak Tw values during a compound drought-heatwave. Johnson and Wilby (2015) reported riparian vegetation yielded the greatest summer cooling (during a non-drought year) effects in upstream regions with modest discharges. The authors highlighted that ~0.5 km of complete shading would reduce July Tw values by 1°C in headwater sites, but ~1.1 km was required in reaches 25 km downstream. Model outputs presented by Jackson et al (2021) supported the effectiveness of such planting strategies on a national-scale across Scotland. Alternatively, in a shaded lowland system in The Netherlands, Kaandorp et al (2019) modelled a riparian clearance scenario and found that maximum summer Tw by ~ 4 °C (scenario 5).

Although fewer studies have tested the empirically tested the effects of riparian clearance and planting on Tw that could support such model outputs, Stott and Marks (2000) represents a notable exception who experimentally tested Tw responses to riparian vegetation clearance in east Wales. The authors reported that summer maximum Tw was 5.3-7.0 °C higher during non-drought conditions following clear-felling practices, despite air temperatures being ~2 °C higher during the 1995 drought in the preceding year (see also O'Driscoll et al., 2016). Similarly, Neal et al (1992) reported that a river system subjected to riparian clearance facilitated maximum summer Tw increases 4–9 °C compared to a shaded reach.

4.2. Flow regulation

Tw responses to flow regulation depends on a multitude of confounding factors, including the reservoir its location in the catchment, inflowing thermal characteristics, water residence times, bathymetry, the potential for thermal stratification, draw off depth and the type of reservoir operation (e.g., water supply, hydropower; Webb and Walling, 1996; 1997; Olden and Naiman, 2010). Scientific evidence on the effects of reservoirs on downstream Tw variations have not been undertaken in a systematic fashion, and inferring generalisable thermal responses is therefore challenging. There remains a fundamental lack of understanding on how different reservoir properties collectively govern Tw, particularly during drought conditions. Various studies from Cfb climates have reported a thermal 'compressing' effect on annual Tw ranges, whereby waters during the summer are cooled by continuous compensation flow releases that restrict the occurrence of extreme low-flows (Webb and Walling 1996; 1997; Krajenbrink et al., 2022). However, various UK studies examining reservoir effects on Tw regimes during droughts have reported mixed findings. In central-northern Wales, Cowx (1987) reported that one regulated system possessed Tw values ~2 °C cooler than a nearby free flowing river during the summer 1976 drought, but another impounded river was ~0.5 °C warmer; the author also reported that the warming or cooling effects of each reservoir were broadly comparable between drought and non-drought years. Jackson et al (2007) highlighted that regulated systems below hydroelectric dams were consistently cooler than non-regulated rivers, respectively reducing inter-annual average and maximum mean daily values by 1–2 °C and 5–6 ° (but the effects of the 2003 compound drought-heatwave were unclear). Webb and Walling (1997) reported that summer Tw values in a regulated system were consistently warmer (up to ~2 °C) than those in a nearby free flowing rivers during the 1989, 1992 and 1995 droughts (albeit less convincingly for the latter) due to the high residence time of impounded waters.

Reservoir destratification measures are the most widely reported means of preventing thermal modifications in regulated systems worldwide (see Olden and Naiman, 2010). Such measures are often introduced to reduce the downstream release of poor water quality conditions (and often colder waters) from the hypolimnion, which can reduce downstream Tw modifications (Cowx et al., 1987; White et al., 2017). Webb and Walling (1997) conducted a detailed study examining the vertical profile of a

reservoir in southwest England where destratification measures are implemented. However, thermal stratification still developed and declining reservoir water levels meant that lower volumes concentrated at deeper cooler offtakes were released downstream, which subsequently lowered Tw for receiving water. Dam removals are another intervention capable of restoring Tw regimes but have been historically limited in the UK due to financial strains, safety concerns in populated areas and the services that reservoirs they provide to society (Habel et al., 2020). However, weir removals are more common in the UK and have the potential to reduce summer thermal peaks during droughts as 'ponded' reaches can warm rapidly (Johnson et al., 2014). Lastly, inter-basin transfer schemes between impounded systems can alter the thermal dynamics of rivers depending on the size and Tw regimes of the donor and receiving waterbodies. For instance, Krajenbrink et al (2022) found that a water transfer scheme reversed the summer cooling effect of a reservoir as the donor basin yielded warmer water temperatures, thus elevating daily mean Tw values by 5 °C during a non-drought summer.

4.3. Water abstraction

Alternative surface water abstraction types than reservoirs include online withdrawals that occur along various river systems, particularly in lowland environments (Cowx, 2000; Dallison et al., 2021). Moreover, groundwater abstraction depicts an invaluable water resource for public water supply across the UK, particularly across the chalk and sandstone aquifers of England and the Carboniferous limestone across various UK regions (Sear et al., 1999). These subsurface withdrawals can limit the input of cool thermal refuges into watercourses during high Tw extremes. Although lowered discharges (and hence likely reduced channel widths and depths) associated with abstraction practices are likely to elevate Tw during drought, this effect is contingent upon water source contributions. For instance, in northeast Scotland Fennell et al (2020) found that Tw variations during the 2018 drought were not sensitive to abstractions and were heavily buffered by cool groundwater inputs. Similarly, on the north island in New Zealand, Dewson et al (2007) experimentally reduced in-channel discharges during summer months to reflect plausible regional abstraction practices and reported small Tw increases due to greater proportional groundwater inputs.

In the UK (and internationally), abstraction volume reductions are most widely implemented to limit the environmental impacts of excessive abstraction during drought. For instance, ‘Hands-off Flow’ (England and Wales) or ‘temporary suspensions’ (Scotland) restrictions enforce license holders to reduce or cease abstraction practices when river discharges fall below a specific threshold, although no research (to our knowledge) has quantified such reductions in abstraction rates on Tw during droughts. However, the lagged effects of groundwater abstraction on river discharges means that such reactive measures are often not feasible (White et al., 2021). Consequently, low-flow alleviation schemes are widely implemented in regions underlain by aquifers, whereby groundwaters are augmented directly into channels when discharges fall below a certain threshold, which can yield cooling effects during extreme low-flows. While this has not been widely researched, Cowx (2000) reported that groundwater augmentation strategies reduced Tw spot measurements by ~0.6 °C during the 1990 drought across a lowland sandstone river system, while Youngs (2017) reported corresponding reductions of 1.5-1.6 °C in a lowland chalk system in a non-drought summer.

4.4. Channelisation

Channel modifications (e.g., for navigation, erosion and flood protection) to channel dimensions have varying implications for Tw variability, but have not been widely explored in the UK. Channel over-deepening can reduce Tw values during drought as solar radiation is attenuated through the water column, as highlighted by Imholt et al (2013) in northeast Scotland during non-drought summer conditions. However, over-deepened channels often yield sluggish flow velocities, and thus experience greater residence times that may facilitate channel warming. Over-widened channels are more likely to increase

Tw values during droughts as such conditions can slow flow velocities (thus enhancing residence time) and expose a greater proportion of the channel to increased solar radiative forcings (Webb and Zhang, 1999; Imholt et al., 2013). Channelisation can also modify Tw dynamics by simplifying hydraulic variations and habitat heterogeneity, and thus limiting hyporheic exchanges between groundwater and surface water (Magliozzi et al., 2019); particularly in urban rivers possessing concrete-lined beds. This may facilitate higher Tw values in channels when cold subsurface water inputs that normally occur in habitats such as riffle tails (Hannah et al., 2009) or groundwater-fed pools (Kaandorp et al., 2019) are absent.

River restoration practices have been widely advocated across the UK and worldwide as a means of reinstating the physical integrity of watercourses and their hydrological and geomorphological processes. However, restoring water quality variables like Tw regimes is rarely a primary motive for such morphological interventions, despite its clear implications. For instance, woody material introductions are a widely recognised and implemented river restoration technique, and Klaar et al (2020) reported how such interventions enhanced streambed temperatures during the summer by enhancing infiltration from the water column. Moreover, restoration techniques like removing paved riverbeds and introducing sinuosity promote morphological variability and thus hyporheic exchanges that has implications for Tw dynamics (Hannah et al., 2009).

5. Modelling river water temperatures under drought

Throughout this review, we have reviewed how various studies across Cfb climate zones have provided a critical understanding of the key processes shaping Tw during hot and dry conditions (i.e., ‘What do we know?’). We have also emphasised a fundamental lack of scientific understanding remains on how different drought properties (e.g., meteorological or hydrological magnitudes and durations) affect the mechanisms governing Tw dynamics (i.e., ‘What don’t we know?’). Novel scientific approaches examining the independent and interactive influences of mechanisms spanning different process sets is urgently required to address such knowledge gaps. Such research should cover various river environments and drought severities (intensities and durations), which could be incorporated within Tw modelling approaches to help guide management initiatives and adaptation strategies aiming to mitigate or offset the effects of high thermal extremes (i.e., ‘What do we need to know?’; Fig. 5). Existing Tw models are predominantly categorised within two classifications: ‘process-based’ or ‘statistical’. The former operates by simulating the real-world transfers of energy and mass that control Tw, while statistical models aim to quantify linkages between Tw and various relevant covariates (for further information, see Benyahya et al., 2007; Dugdale et al. 2017). In this section, we highlight and discuss 5 critical research questions surrounding modelling approaches that need to be addressed to better predict drought-induced Tw dynamics.

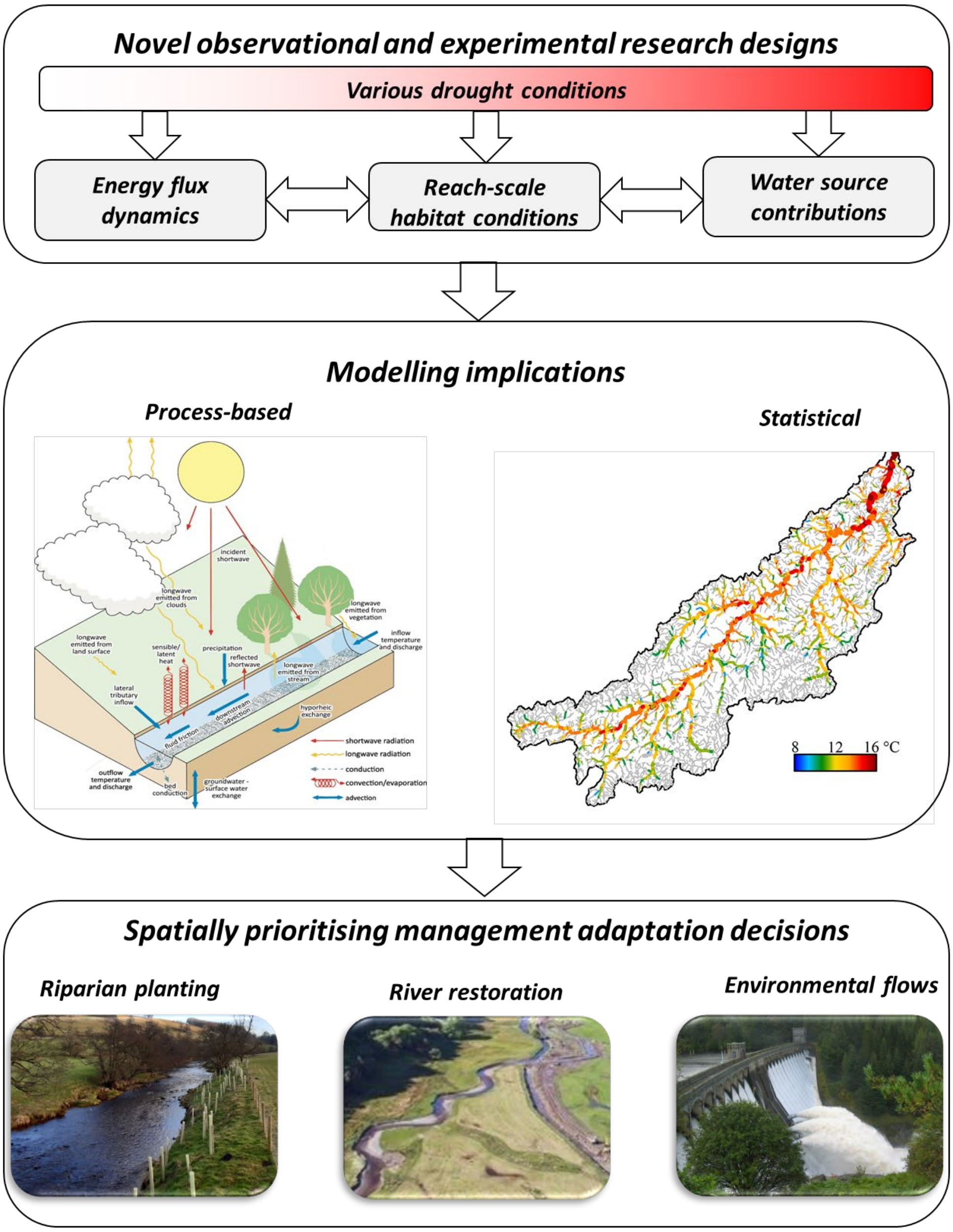


Fig. 5 – A schematic flow chart indicating different types of new scientific evidence is required to answer critical research questions surrounding Tw dynamics during droughts and how this could translate to spatially prioritising effective management adaptation decisions. Photo credit: 'Process-based model' – Fig. 2 in Dugdale et al (2018); 'Statistical model' – Fig. 4 in Jackson et al (2017); 'Riparian planting' – Ribbles Trust, 2021; 'River restoration' – FBA, 2023; 'Environmental flows' – Jennifer Jones, 2022).

5.1. Which present day and future hydroclimatic conditions characterising drought should be modelled?

Although some studies detailing process-based (e.g., Garner et al., 2014; 2017) and statistical Tw models (e.g., van Vliet et al., 2011; Beaufort et al., 2022) have incorporated high air temperatures and extreme low-flow conditions, such approaches often focus on seasonally typical summer conditions (e.g., Jackson et al., 2018; 2021; Kaandorp et al., 2019). This represents a major limitation for all Tw models as drought-induced Tw dynamic predictions would be beyond the calibration range of the training data collected during non-drought conditions. For process-based models, many of the underpinning energy influences have been largely untested during extreme droughts, including the performance of empirical or semi-empirical approaches widely employed used to estimate latent heat fluxes (e.g., Garner et al., 2014; 2017). Solar radiation receipts incorporated within large-scale statistical models will experience greater uncertainty for this as the underpinning energy flux estimates are derived from coarser spatial scales (e.g., Johnson and Wilby, 2015; Jackson et al., 2021). Consequently, further empirical evidence and modelling studies are required during drought conditions that can better parameterise the effects of different processes shaping Tw dynamics. Regarding future hydroclimatic conditions, scenarios indicating air temperature, precipitation and river flow shifts are consistently outputted within climate change forecasts (e.g., Lowe et al., 2019). Changes in thermal energy budget controls like solar radiation is less common, although can be estimated based on forecasted changes to sunlight hour predictions (Burnett et al., 2014). However, caution should be exercised when assuming stationarity in drought conditions, as is widely applied within Tw models (particularly statistical models), given that the duration and severity of such events are likely to increase in the future (Yin et al., 2022). For instance,

5.2. How can hydraulic conditions reflecting drought be incorporated into Tw models?

Various process-based models include a hydraulic routing component (e.g., HEC-RAS - Saleh et al. 2013), which often use a formulation of the St-Venant equations to simulate flow velocity and depth. However, under extreme low-flow scenarios where the bed roughness height approaches the water depth, stable solutions to these equations can be difficult to achieve (e.g., Saleh et al. 2013), meaning that such models often have difficulty accurately simulating water velocity and depth, potentially leading to inaccurate estimation of resulting Tw. The use of such hydraulic models have been most widely utilised within North America and often tailored towards high-flow events (Dugdale et al., 2017). Given the importance of hydraulic conditions for Tw, process-based models incorporating hydraulic routing components will continue to be a fundamental tool for quantifying river thermal extremes, but such conditions should be more widely adapted and parameterised to extreme low-flow conditions and need require further testing in Cfb climates. For statistical models spanning large spatial scales, incorporating hydraulic geometry influences is far more challenging (Benyahya et al., 2007). Estimates of channel slope and width can be derived from GIS information (e.g., Jackson et al., 2021), but hydrological information is also required (see below) to estimate hydraulic responses to changing flow conditions, which can be biased by velocity-discharge relationships exhibited at flow gauging stations (which often yield unnatural river cross-sections to facilitate reliable discharge measurements).

5.3. How can hydrological and Tw models be more effectively synergised to predict thermal responses to drought?

Hydrological data is most widely available via flow gauges that can provide robust river discharge timeseries but are spatially discrete in nature. This vastly restricts the quality of underpinning training data by limiting the number of paired temperature and discharge loggers, and also constrains large-scale spatial Tw predictions as hydrological estimates can be difficult to model continuously across river networks (but see below). Hydrological models can provide an alternative means of deriving spatially continuous river discharge data, but are often only available up to regional scales and can face various challenges predicting processes like discharge variations, groundwater disconnections and flow cessation events (e.g., White et al., 2018) during droughts (see Smith et al., 2019). This is a particular knowledge gap as recognising where different water source disconnections occur, either through upstream advective inputs (i.e., drying events or instream ponding) or groundwater contributions is critical for recognising Tw shifts during drought conditions. However, some hydrological models can help identify the effects of different water source contributions on Tw during drought conditions. For instance, during a non-drought year in northeast Scotland, Fabris et al (2018) combined process-based Tw and hydrological ('MIKE 11') models to predict the effects of both surface water and groundwater contributions on river thermal properties alongside hydraulic influences and riparian plant coverages. Various statistical Tw models have incorporated discharge estimates, most commonly for studies aiming to utilise flow magnitudes as a covariate alongside air temperature to predict Tw (e.g., Webb et al., 2003; Van Vliet et al., 2011). However, most of such studies depict spatially discrete relationships due to available flow gauge data (see above). Statistical models estimating spatially continuous high Tw often quantify the effects of upstream advective inputs via stream order as a surrogate for river discharges (e.g., Beaufort et al., 2016; Jackson et al., 2017; 2018), which although practical at large spatial scales overlooks the nuances of flow regime variations.

5.4. How can human activity influences on Tw be quantified and modelled?

Unsurprisingly, riparian planting or clearance depict the human activities most widely modelled in relation to Tw, which has been most widely explored within process-based models that can capture the effects of channel shading (see Section 4.1). Process-based Tw models can more readily incorporate the influences of different human activities (e.g., dam-induced hydrological modifications – Buddendorf et al., 2017; Wawrzyniak et al., 2017) by parameterising their effects operating within a single system. Accounting for hydraulic and hydrological conditions (described above) modified could be more widely incorporated within Tw models. This becomes more challenging for statistical models spanning large spatial scales as the effects of human modifications like dams and channel modifications can vary within and between river catchments. However, spatially continuous estimates of morphological pressures or sub-reach flow properties (Naura et al., 2018; Magliozzi et al., 2019) could help characterise hydraulic geometries that mediate channel velocities and width:depth ratios. Moreover, hydrological models can provide a measure of surface and subsurface water management influences on river discharges across river networks (White et al., 2018; 2021), which could also help refine and identify various future drought scenarios. For instance, in a global study, Wanders et al (2019) found that accounting for water management operations like flow regulation and water abstractions improved Tw model predictions. The hydrological and thermal impacts of reservoirs are difficult to model.

5.5. How can we more effectively model and spatially prioritise the effects of different management interventions?

Computational advances allowing solar radiation receipts to be quantified across river grid cells globally can afford process-based models to be undertaken across larger spatial scales and hence help guide management actions. Jackson et al (2021) provided a novel, simplified process-based Tw modelling approach that allowed estimates of riparian shading, solar radiation receipt, river discharge, hydraulic conditions (residence times) and an array of landscape and channel characteristics to be projected across large spatial scales. From this, they derived a planting prioritisation metric to indicate where afforestation would likely yield the greatest reductions in incoming radiation and summer Tw values. Although various other catchment-wide, spatial prioritisation examples have been derived based on process-based Tw models (e.g., Collier et al., 2001; Johnson and Wilby, 2015), such examples have been most consistently related to typical summer low-flow conditions and not tailored to drought conditions. Although statistical Tw models have been widely utilised to identify locations vulnerable to high thermal extremes during low-flow conditions at up national scales (e.g., Jackson et al., 2018; Beaufort et al., 2022), such approaches have not been as widely incorporated within spatial prioritisation approaches. However, both types Tw models have not been widely used to highlight and spatially prioritise alternative management approaches like river restoration strategies or environmental flows, but building in metrics of morphological pressures or hydraulic geometries and hydrological alterations (outlined above) could respectively help prioritise such interventions.

6. Recommendations

We conclude by outlining four key recommendations to help better understand, predict and adapt to drought-induced Tw dynamics based on information gleaned from this review.

Processes – Our scientific thinking around river temperature extremes during drought (and heatwaves) needs to be re-conceptualised so that **energy flux dynamics, reach-scale habitat** (hydraulic and riparian) influences and **water source contributions** (advective flows and heat) are more widely considered.

Data - **Large-scale, high frequency river temperature monitoring networks** spanning different environmental gradients are required to more accurately model river temperature dynamics during future droughts and at unmonitored sites (although this would require long-term government investment). Drought effects of different **Tw drivers spanning multiple process sets** are also needed across **various river environments** to better support model predictions.

Models - **Hydrological information that better characterise drought conditions** should be more widely incorporated within river temperature models that are often based on climate and landscape controls.

Decision making - An **evidence-based** approach that utilises river temperature data and models is required to **better inform management interventions** capable of mitigating high temperature extremes.

References

AghaKouchak, A., Chiang, F., Huning, L.S., Love, C.A., Mallakpour, I., Mazdiyasni, O., Moftakhari, H., Papalexiou, S.M., Ragno, E. & Sadegh, M. (2020) 'Climate extremes and compound hazards in a warming world', Annual Review of Earth and Planetary Sciences, 48(1), pp. 519–548.

Arias Font, R., Khamis, K., Milner, A.M., Sambrook Smith, G.H. & Ledger, M.E. (2021) 'Low flow and heatwaves alter ecosystem functioning in a stream mesocosm experiment', Science of The Total Environment, 777p. 146067.

Bachiller-Jareno, N., Hutchins, M.G., Bowes, M.J., Charlton, M.B. & Orr, H.G. (2019) 'A novel application of remote sensing for modelling impacts of tree shading on water quality', Journal of Environmental Management, 230pp. 33–42.

Barker, L.J., Hannaford, J., Parry, S., Smith, K.A., Tanguy, M. & Prudhomme, C. (2019) 'Historic hydrological droughts 1891–2015: systematic characterisation for a diverse set of catchments across the UK', Hydrology and Earth System Sciences, 23(11), pp. 4583–4602.

Beaufort, A., Diamond, J.S., Sauquet, E. & Moatar, F. (2022) 'Spatial extrapolation of stream thermal peaks using heterogeneous time series at a national scale', Hydrology and Earth System Sciences, 26(13), pp. 3477–3495.

Beaufort, A., Moatar, F., Curie, F., Ducharne, A., Bustillo, V. & Thiéry, D. (2016) 'River Temperature Modelling by Strahler Order at the Regional Scale in the Loire River Basin, France', River Research and Applications, 32(4), pp. 597–609.

Beckett, A.D. & Sanderson, M.G. (2022) 'Analysis of historical heatwaves in the United Kingdom using gridded temperature data', International Journal of Climatology, 42(1), pp. 453–464.

Bell, V.A., Naden, P.S., Tipping, E., Davies, H.N., Carnell, E., Davies, J.A.C., Dore, A.J., Dragosits, U., Lapworth, D.J., Muhammed, S.E., Quinton, J.N., Stuart, M., Tomlinson, S., Wang, L., Whitmore, A.P. & Wu, L. (2021) 'Long term simulations of macronutrients (C, N and P) in UK freshwaters', Science of The Total Environment, 776p. 145813.

Benyahya, L., Caissie, D., St-Hilaire, A., Ouarda, T.B.M.J. & Bobée, B. (2007) 'A Review of Statistical Water Temperature Models', Canadian Water Resources Journal / Revue canadienne des ressources hydriques, 32(3), pp. 179–192.

Booker, D.J. & Whitehead, A.L. (2022) 'River water temperatures are higher during lower flows after accounting for meteorological variability', River Research and Applications, 38(1), pp. 3–22.

Bowes, M.J., Smith, J.T., Neal, C., Leach, D.V., Scarlett, P.M., Wickham, H.D., Harman, S.A., Armstrong, L.K., Davy-Bowker, J., Haft, M. & Davies, C.E. (2011) 'Changes in water quality of the River Frome (UK) from 1965 to 2009: Is phosphorus mitigation finally working?', Science of The Total Environment, 409(18), pp. 3418–3430.

Broadmeadow, S.B., Jones, J.G., Langford, T.E.L., Shaw, P.J. & Nisbet, T.R. (2011) 'The influence of riparian shade on lowland stream water temperatures in southern England and their viability for brown trout', River Research and Applications, 27(2), pp. 226–237.

Brooker, M.P., Morris, D.L. & Hemsworth, R.J. (1977) 'Mass Mortalities of Adult Salmon, Salmo salar, in the R. Wye, 1976', Journal of Applied Ecology, 14(2), pp. 409–417.

Buddendorf, W.B., Malcolm, I.A., Geris, J., Fabris, L., Millidine, K.J., Wilkinson, M.E. & Soulsby, C. (2017) 'Spatio-temporal effects of river regulation on habitat quality for Atlantic salmon fry', Ecological Indicators, 83pp. 292–302.

Burnett, D., Barbour, E. & Harrison, G.P. (2014) 'The UK solar energy resource and the impact of climate change', Renewable Energy, 71pp. 333–343.

Caruso, B., S. (2001) 'Regional river flow, water quality, aquatic ecological impacts and recovery from drought', Hydrological Sciences Journal, 46(5), pp. 677–699.

Collier, K.J., Rutherford, J.C., Quinn, J.M. & Davies-Colley, R.J. (2001) 'Forecasting rehabilitation outcomes for degraded New Zealand pastoral streams', Water Science and Technology, 43(9), pp. 175–184.

Cooper, R.J., Hiscock, K.M., Lovett, A.A., Dugdale, S.J., Sünnenberg, G. & Vrain, E. (2020) 'Temporal hydrochemical dynamics of the River Wensum, UK: Observations from long-term high-resolution monitoring (2011–2018)', Science of The Total Environment, 724p. 138253.

Cowx, I.G. (2000) 'Potential impact of groundwater augmentation of river flows on fisheries: a case study from the River Ouse, Yorkshire, UK', Fisheries Management and Ecology, 7(1–2), pp. 85–96.

Cowx, I.G., Young, W.O. & Booth, J.P. (1987) 'Thermal characteristics of two regulated rivers in mid-wales, U.K.', Regulated Rivers: Research & Management, 1(1), pp. 85–91.

Crisp, D.T. (1997) 'Water temperature of Plynlimon streams', Hydrology and Earth System Sciences, 1(3), pp. 535–540.

Crisp, D.T., Matthews, A.M., Westlake, D.F., & Freshwater Biological Association (1982) 'The temperatures of nine flowing waters in southern England', Hydrobiologia, 89(3), pp. 193–204.

Dallison, R.J.H., Patil, S.D. & Williams, A.P. (2021) 'Impacts of climate change on future water availability for hydropower and public water supply in Wales, UK', Journal of Hydrology: Regional Studies, 36p. 100866.

Datry, T. (2017) 'Ecological Effects of Flow Intermittence in Gravel-Bed Rivers', in Gravel-Bed Rivers. [Online]. John Wiley & Sons, Ltd. pp. 261–297.

Dewson, Z.S., James, A.B.W. & Death, R.G. (2007) 'Invertebrate responses to short-term water abstraction in small New Zealand streams', Freshwater Biology, 52(2), pp. 357–369.

Dugdale, S.J., Hannah, D.M. & Malcolm, I.A. (2017) ‘River temperature modelling: A review of process-based approaches and future directions’, Earth-Science Reviews, 175pp. 97–113.

Dugdale, S.J., Malcolm, I.A., Kantola, K. & Hannah, D.M. (2018) ‘Stream temperature under contrasting riparian forest cover: Understanding thermal dynamics and heat exchange processes’, Science of The Total Environment, 610–611pp. 1375–1389.

Elliott, J.M. (2000) ‘Pools as refugia for brown trout during two summer droughts: trout responses to thermal and oxygen stress’, Journal of Fish Biology, 56(4), pp. 938–948.

Elliott, J.M. & Elliott, J.A. (2010) ‘Temperature requirements of Atlantic salmon Salmo salar, brown trout Salmo trutta and Arctic charr Salvelinus alpinus: predicting the effects of climate change’, Journal of Fish Biology, 77(8), pp. 1793–1817.

Environment Agency (2023) ‘Hydrology data explorer’. Available online via https://environment.data.gov.uk/hydrology [Accessed 31/3/2023].

Evans, E.C., McGregor, G.R. & Petts, G.E. (1998) ‘River energy budgets with special reference to river bed processes’, Hydrological Processes, 12(4), pp. 575–595.

Fabris, L., Malcolm, I.A., Buddendorf, W.B. & Soulsby, C. (2018) ‘Integrating process-based flow and temperature models to assess riparian forests and temperature amelioration in salmon streams’, Hydrological Processes, 32(6), pp. 776–791.

Fennell, J., Geris, J., Wilkinson, M.E., Daalmans, R. & Soulsby, C. (2020) ‘Lessons from the 2018 drought for management of local water supplies in upland areas: A tracer-based assessment’, Hydrological Processes, 34(22), pp. 4190–4210.

Ferguson, R.I. (1986) ‘Hydraulics and hydraulic geometry’, Progress in Physical Geography: Earth and Environment, 10(1), pp. 1–31.

Ficklin, D.L., Hannah, D.M., Wanders, N., Dugdale, S.J., England, J., Klaus, J., Kelleher, C., Khamis, K. & Charlton, M.B. (2023) ‘Rethinking river water temperature in a changing, human-dominated world’, Nature Water, 1(2), pp. 125–128.

Folegot, S., Hannah, D.M., Dugdale, S.J., Kurz, M.J., Drummond, J.D., Klaar, M.J., Lee-Cullin, J., Keller, T., Martí, E., Zarnetske, J.P., Ward, A.S. & Krause, S. (2018) ‘Low flow controls on stream thermal dynamics’, Limnologica, 68pp. 157–167.

Freshwater Biological Association (2023) ‘River restoration training’. Available online via https://www.fba.org.uk/shop/p/river-restoration-training [Accessed 03/04/2023].

Garner, G., Malcolm, I.A., Sadler, J.P. & Hannah, D.M. (2017) ‘The role of riparian vegetation density, channel orientation and water velocity in determining river temperature dynamics’, Journal of Hydrology, 553pp. 471–485.

Garner, G., Malcolm, I.A., Sadler, J.P. & Hannah, D.M. (2014) ‘What causes cooling water temperature gradients in a forested stream reach?’, Hydrology and Earth System Sciences, 18(12), pp. 5361–5376.

Garner, G., Malcolm, I.A., Sadler, J.P., Millar, C.P. & Hannah, D.M. (2015) 'Inter-annual variability in the effects of riparian woodland on micro-climate, energy exchanges and water temperature of an upland Scottish stream', Hydrological Processes, 29(6), pp. 1080–1095.

Hannah, D.M. & Garner, G. (2015) 'River water temperature in the United Kingdom: Changes over the 20th century and possible changes over the 21st century', Progress in Physical Geography: Earth and Environment, 39(1), pp. 68–92.

Hannah, D.M., Malcolm, I.A. & Bradley, C. (2009) 'Seasonal hyporheic temperature dynamics over riffle bedforms', Hydrological Processes, 23(15), pp. 2178–2194.

Hannah, D.M., Malcolm, I.A., Soulsby, C. & Youngson, A.F. (2008) 'A comparison of forest and moorland stream microclimate, heat exchanges and thermal dynamics', Hydrological Processes, 22(7), pp. 919–940.

Hannah, D.M., Malcolm, I.A., Soulsby, C. & Youngson, A.F. (2004) 'Heat exchanges and temperatures within a salmon spawning stream in the Cairngorms, Scotland: seasonal and sub-seasonal dynamics', River Research and Applications, 20(6), pp. 635–652.

Hrachowitz, M., Soulsby, C., Imholt, C., Malcolm, I.A. & Tetzlaff, D. (2010) 'Thermal regimes in a large upland salmon river: a simple model to identify the influence of landscape controls and climate change on maximum temperatures', Hydrological Processes, 24(23), pp. 3374–3391.

Hutchins, M.G., Williams, R.J., Prudhomme, C., Bowes, M.J., Brown, H.E., Waylett, A.J. & Loewenthal, M. (2016) 'Projections of future deterioration in UK river quality are hampered by climatic uncertainty under extreme conditions', Hydrological Sciences Journal, 61(16), pp. 2818–2833.

Ilyas, M., Nisar, M., Khan, N., Hazrat, A., Khan, A.H., Hayat, K., Fahad, S., Khan, A. & Ullah, A. (2021) 'Drought Tolerance Strategies in Plants: A Mechanistic Approach', Journal of Plant Growth Regulation, 40(3), pp. 926–944.

Imholt, C., Soulsby, C., Malcolm, I. a. & Gibbins, C. n. (2013) 'Influence of contrasting riparian forest cover on stream temperature dynamics in salmonid spawning and nursery streams', Ecohydrology, 6(3), pp. 380–392.

Jackson, F. l., Hannah, D.M., Fryer, R. j., Millar, C. p. & Malcolm, I. a. (2017) 'Development of spatial regression models for predicting summer river temperatures from landscape characteristics: Implications for land and fisheries management', Hydrological Processes, 31(6), pp. 1225–1238.

Jackson, F.L., Fryer, R.J., Hannah, D.M., Millar, C.P. & Malcolm, I.A. (2018) 'A spatio-temporal statistical model of maximum daily river temperatures to inform the management of Scotland's Atlantic salmon rivers under climate change', Science of The Total Environment, 612pp. 1543–1558.

Jackson, F.L., Hannah, D.M., Ouellet, V. & Malcolm, I.A. (2021) ‘A deterministic river temperature model to prioritize management of riparian woodlands to reduce summer maximum river temperatures’, Hydrological Processes, 35(8), p. e14314.

Jackson, H.M., Gibbins, C.N. & Soulsby, C. (2007) ‘Role of discharge and temperature variation in determining invertebrate community structure in a regulated river’, River Research and Applications, 23(6), pp. 651–669.

Jennifer Jones (2022) ‘Laggan Dam in full flow - geograph.org.uk - 3006049.jpg’. Available online via https://commons.wikimedia.org/wiki/File:Laggan_Dam_in_full_flow_-_geograph.org.uk_-_3006049.jpg

Johnson, M.F. & Wilby, R.L. (2015) ‘Seeing the landscape for the trees: Metrics to guide riparian shade management in river catchments’, Water Resources Research, 51(5), pp. 3754–3769.

Johnson, M.F., Wilby, R.L. & Toone, J.A. (2014) ‘Inferring air–water temperature relationships from river and catchment properties’, Hydrological Processes, 28(6), pp. 2912–2928.

Kaandorp, V.P., Doornenbal, P.J., Kooi, H., Peter Broers, H. & de Louw, P.G.B. (2019) ‘Temperature buffering by groundwater in ecologically valuable lowland streams under current and future climate conditions’, Journal of Hydrology X, 3p. 100031.

Krajenbrink, H.J., White, J.C., Dunbar, M.J. & Wood, P.J. (2022) ‘Macroinvertebrate and diatom community responses to thermal alterations below water supply reservoirs’, River Research and Applications, 38(3), pp. 595–612.

Laizé, C.L.R., Bruna Meredith, C., Dunbar, M.J. & Hannah, D.M. (2017) ‘Climate and basin drivers of seasonal river water temperature dynamics’, Hydrology and Earth System Sciences, 21(6), pp. 3231–3247.

Langan, S.J., Johnston, L., Donaghy, M.J., Youngson, A.F., Hay, D.W. & Soulsby, C. (2001) ‘Variation in river water temperatures in an upland stream over a 30-year period’, Science of The Total Environment, 265(1), pp. 195–207.

Leach, J.A., Kelleher, C., Kurylyk, B.L., Moore, R.D. & Neilson, B.T. (n.d.) ‘A primer on stream temperature processes’, WIREs Water, n/a(n/a), p. e1643.

Liu, S., Xie, Zhenghui, Liu, B., Wang, Y., Gao, J., Zeng, Y., Xie, J., Xie, Zhipeng, Jia, B., Qin, P., Li, R., Wang, L. & Chen, S. (2020) ‘Global river water warming due to climate change and anthropogenic heat emission’, Global and Planetary Change, 193p. 103289.

Lowe, J. A., Bernie, D., Bett, P., Bricheno, L., Brown, S., Calvert, D., Clark, R., Eagle, K., Edwards, T., Fosser, G., and Fung, F. (2019). ‘UKCP18 science overview report’. Met Office Hadley Centre: Exeter, UK.

Lu, X., Gray, C., Brown, L.E., Ledger, M.E., Milner, A.M., Mondragón, R.J., Woodward, G. & Ma, A. (2016) ‘Drought rewires the cores of food webs’, Nature Climate Change, 6(9), pp. 875–878.

Magliozzi, C., Coro, G., Grabowski, R.C., Packman, A.I. & Krause, S. (2019) 'A multiscale statistical method to identify potential areas of hyporheic exchange for river restoration planning', Environmental Modelling & Software, 111pp. 311–323.

Merritt, D.M., Scott, M.L., LeROY POFF, N., Auble, G.T. & Lytle, D.A. (2010) 'Theory, methods and tools for determining environmental flows for riparian vegetation: riparian vegetation-flow response guilds', Freshwater Biology, 55(1), pp. 206–225.

Naura, M., Clark, M.J., Sear, D.A., Atkinson, P.M., Hornby, D.D., Kemp, P., England, J., Peirson, G., Bromley, C. & Carter, M.G. (2016) 'Mapping habitat indices across river networks using spatial statistical modelling of River Habitat Survey data', Ecological Indicators, 66pp. 20–29.

Neal, C, Smith, C. J, Hill, S and (1992) 'Forestry impact on upland water quality'. Institute of Hydrology Report No. 119. Available online via https://nora.nerc.ac.uk/id/eprint/7364/1/IH_119.pdf [Accessed 03/04/2023]

O'Driscoll, C., O'Connor, M., Asam, Z.-Z., de Eyto, E., Brown, L.E. & Xiao, L. (2016) 'Forest clearfelling effects on dissolved oxygen and metabolism in peatland streams', Journal of Environmental Management, 166pp. 250–259.

Olden, J.D. & Naiman, R.J. (2010) 'Incorporating thermal regimes into environmental flows assessments: modifying dam operations to restore freshwater ecosystem integrity', Freshwater Biology, 55(1), pp. 86–107.

Orr, H.G., Johnson, M.F., Wilby, R.L., Hatton-Ellis, T. & Broadmeadow, S. (2015) 'What else do managers need to know about warming rivers? A United Kingdom perspective', WIREs Water, 2(2), pp. 55–64.

Ouellet, V., St-Hilaire, A., Dugdale, S.J., Hannah, D.M., Krause, S. & Proulx-Ouellet, S. (2020) 'River temperature research and practice: Recent challenges and emerging opportunities for managing thermal habitat conditions in stream ecosystems', Science of The Total Environment, 736p. 139679.

Ribble Rivers Trust (2021) 'Rivers and trees - 5 benefits for woodlands and water'. Available online via https://ribbletrust.org.uk/rivers-and-trees-five-benefits-for-woodlands-and-water/ [Accessed 03/04/2023].

Saleh, F., Ducharne, A., Flipo, N., Oudin, L. & Ledoux, E. (2013) 'Impact of river bed morphology on discharge and water levels simulated by a 1D Saint–Venant hydraulic model at regional scale', Journal of Hydrology, 476pp. 169–177.

Sear, D.A., Armitage, P.D. & Dawson, F.H. (1999) 'Groundwater dominated rivers', Hydrological Processes, 13(3), pp. 255–276.

Sherwood, S.C., Dixit, V. & Salomez, C. (2018) 'The global warming potential of near-surface emitted water vapour', Environmental Research Letters, 13(10), p. 104006.

Stott, T. & Marks, S. (2000) ‘Effects of plantation forest clearfelling on stream temperatures in the Plynlimon experimental catchments, mid-Wales’, Hydrology and Earth System Sciences, 4(1), pp. 95–104.

Turner, S., Barker, L.J., Hannaford, J., Muchan, K., Parry, S. & Sefton, C. (2021) ‘The 2018/2019 drought in the UK: a hydrological appraisal’, Weather, 76(8), pp. 248–253.

Van Loon, A.F., Gleeson, T., Clark, J., Van Dijk, A.I.J.M., Stahl, K., Hannaford, J., Di Baldassarre, G., Teuling, A.J., Tallaksen, L.M., Uijlenhoet, R., Hannah, D.M., Sheffield, J., Svoboda, M., Verbeiren, B., Wagener, T., Rangecroft, S., Wanders, N. & Van Lanen, H.A.J. (2016) ‘Drought in the Anthropocene’, Nature Geoscience, 9(2), pp. 89–91.

Verberk, W.C.E.P., Durance, I., Vaughan, I.P. & Ormerod, S.J. (2016) ‘Field and laboratory studies reveal interacting effects of stream oxygenation and warming on aquatic ectotherms’, Global Change Biology, 22(5), pp. 1769–1778.

van Vliet, M.T.H., Ludwig, F., Zwolsman, J.J.G., Weedon, G.P. & Kabat, P. (2011) ‘Global river temperatures and sensitivity to atmospheric warming and changes in river flow’, Water Resources Research, 47(2), .

van Vliet, M.T.H. & Zwolsman, J.J.G. (2008) ‘Impact of summer droughts on the water quality of the Meuse river’, Journal of Hydrology, 353(1), pp. 1–17.

Wanders, N., van Vliet, M.T.H., Wada, Y., Bierkens, M.F.P. & van Beek, L.P.H. (Rens) (2019) ‘High-Resolution Global Water Temperature Modeling’, Water Resources Research, 55(4), pp. 2760–2778.

Wawrzyniak, V., Allemand, P., Bailly, S., Lejot, J. & Piégay, H. (2017) ‘Coupling LiDAR and thermal imagery to model the effects of riparian vegetation shade and groundwater inputs on summer river temperature’, Science of The Total Environment, 592pp. 616–626.

Webb, B.W., Clack, P.D. & Walling, D.E. (2003) ‘Water–air temperature relationships in a Devon river system and the role of flow’, Hydrological Processes, 17(15), pp. 3069–3084.

Webb, B.W. & Crisp, D.T. (2006) ‘Afforestation and stream temperature in a temperate maritime environment’, Hydrological Processes, 20(1), pp. 51–66.

Webb, B.W. & Walling, D.E. (1997) ‘Complex summer water temperature behaviour below a UK regulating reservoir’, Regulated Rivers: Research & Management, 13(5), pp. 463–477.

Webb, B.W. & Walling, D.E. (1993) ‘Longer-term water temperature behaviour in an upland stream’, Hydrological Processes, 7(1), pp. 19–32.

Webb, B.W. & Zhang, Y. (2004) ‘Intra-annual variability in the non-advective heat energy budget of Devon streams and rivers’, Hydrological Processes, 18(11), pp. 2117–2146.

Webb, B.W. & Zhang, Y. (1997) ‘Spatial and seasonal variability in the components of the river heat budget’, Hydrological Processes, 11(1), pp. 79–101.

Webb, B.W. & Zhang, Y. (1999) 'Water temperatures and heat budgets in Dorset chalk water courses', Hydrological Processes, 13(3), pp. 309–321.

Westhoff, M.C., Savenije, H.H.G., Luxemburg, W.M.J., Stelling, G.S., van de Giesen, N.C., Selker, J.S., Pfister, L. & Uhlenbrook, S. (2007) 'A distributed stream temperature model using high resolution temperature observations', Hydrology and Earth System Sciences, 11(4), pp. 1469–1480.

White, J.C., Aspin, T.W.H., Picken, J.L., Ledger, M.E., Wilby, R.L. & Wood, P.J. (2022) 'Extreme low-flow effects on riverine fauna: A perspective on methodological assessments', Ecohydrology, 15(5), p. e2422.

White, J.C., Fornaroli, R., Hill, M.J., Hannah, D.M., House, A., Colley, I., Perkins, M. & Wood, P.J. (2021) 'Long-term river invertebrate community responses to groundwater and surface water management operations', Water Research, 189p. 116651.

White, J.C., Hannah, D.M., House, A., Beatson, S.J.V., Martin, A. & Wood, P.J. (2017) 'Macroinvertebrate responses to flow and stream temperature variability across regulated and non-regulated rivers', Ecohydrology, 10(1), p. e1773.

White, J.C., House, A., Punchard, N., Hannah, D.M., Wilding, N.A. & Wood, P.J. (2018) 'Macroinvertebrate community responses to hydrological controls and groundwater abstraction effects across intermittent and perennial headwater streams', Science of The Total Environment, 610–611pp. 1514–1526.

Wilby, R.L. & Johnson, M.F. (2020) 'Climate variability and implications for keeping rivers cool in England', Climate Risk Management, 30p. 100259.

Wilby, Robert L., Johnson, M.F. & Toone, J.A. (2015) 'Thermal shockwaves in an upland river', Weather, 3(70), pp. 92–100.

Wilby, R. L., Prudhomme, C., Parry, S. & Muchan, K.G.L. (2015) 'Persistence of Hydrometeorological Droughts in the United Kingdom: A Regional Analysis of Multi-Season Rainfall and River Flow Anomalies', Journal of Extreme Events, 02(02), p. 1550006.

Wood, P.J., Gunn, J., Smith, H. & Abas-Kutty, A. (2005) 'Flow permanence and macroinvertebrate community diversity within groundwater dominated headwater streams and springs', Hydrobiologia, 545(1), pp. 55–64.

Yin, J., Slater, L., Gu, L., Liao, Z., Guo, S. & Gentine, P. (2022) 'Global Increases in Lethal Compound Heat Stress: Hydrological Drought Hazards Under Climate Change', Geophysical Research Letters, 49(18), p. e2022GL100880.

Youngs, M (2017) 'An assessment of the ecological impact of a groundwater abstraction stream support system on macroinvertebrate community structure in the River Piddle, Dorset, UK' MSc thesis, Loughborough University, Leicestershire, UK.

Zwolsman, J.J.G. & van Bokhoven, A.J. (2007) ‘Impact of summer droughts on water quality of the Rhine River - a preview of climate change?’, Water Science and Technology, 56(4), pp. 45–55.

G: The ecological effects of drought on England’s rivers

Prof. Rachel Stubbington, Nottingham Trent University

Overview

Summary

This review considers the ecological effects of drought—which may manifest instream as conditions from low flows to long dry phases—in England’s rivers and streams. The review:

- focuses on aquatic invertebrates, due to their abundance, biodiversity, ecological importance, ubiquity, well-known habitat preferences—and thus their well-documented responses to drought. Other communities, including microorganisms, plants and fish, are also considered.
- includes all English river and stream types as represented in the literature, and as thus is somewhat biased towards consideration of chalk streams in south England.
- includes historic as well as recent droughts, but not the 2022-onset drought, the ecological effects of which have yet to be reported.

Key findings

What do we know? We know that:

- The biodiversity and biomass of groups including plants, invertebrates and fish decrease during drought. Streambed drying causes severe declines and losses of aquatic species.
- Key species including habitat-forming plants such as water crowfoot (*Ranunculus*) and salmonid fish are among those at risk of population decline and loss.
- Post-drought recovery can be rapid—but some species take years to return. Moreover, reports of rapid recovery are often snapshots that lack pre-drought data, fail to track recovery to completion, and characterize communities already shaped by disturbances.
- Communities in relatively natural rivers with diverse refuges are more resilient to drought than those exposed to greater human impacts.

What don’t we know?

- We know that drought could push ecosystems past tipping points, causing shifts to alternative states that are hard to reverse. We need to learn how to identify systems on the approach to a tipping point, to inform timely and appropriate management actions.
- We know most about local, site-specific responses to drought, whereas we know little about how local responses collectively determine regional effects of drought.
- Our limited understanding of responses to past drought is insufficient to inform robust predictions of how increases in drought severity will interact with other climatic and human pressures to shape ecological responses to future events.

Recommendations

What do we need to do?

Management actions can promote ecological resilience to drought. In particular, we need to:

- **protect and/or restore natural flow regimes** in particular to stop drying of naturally perennial rivers, and to maintain flows needed, for example, by migratory fish.
- **restore natural processes** to promote development of riparian habitats and channel shapes that include drought refuges, such as deep pools and accessible bed sediments.
- **improve water quality** alongside the riparian and channel habitats, to enhance resilience to drought and other climatic stressors that we cannot control.
- **take action to re-establish populations** of key species eliminated by drought.

1. Non-technical summary

Drought is no longer entirely natural. Although drought is fundamentally natural, it may be changing in frequency and severity due to climate change. In addition, rivers experience drought alongside a range of human pressures including water resource use, exacerbating its ecological impacts. As such, drought is no longer a purely natural phenomenon.

Ecological responses to drought vary. Responses vary depending on how drought alters habitats, and vary among rivers, and among habitats in each river. Responses vary among communities, among taxa in a community, among species in a genus, and among individuals within a species. This variability should be borne in mind when considering general patterns.

Drought increases the risk of local to national-scale species extinctions. Drought can locally eliminate a species, and this risk increases with both drought magnitude and duration. Depending on whether lost species recolonize from populations surviving in refuges, local losses could accumulate and ultimately lead to national extinction.

Drought and heatwaves interact to exacerbate extinction risk. Warming waters are putting temperature-sensitive species—which are already restricted to waters meeting their thermal requirements—at risk of extinction. Drought reduces water depths, increasing the influence of solar radiation and thus the risk of temperatures that eliminate populations.

Recovery after past droughts may not represent recovery after future droughts. Evidence suggests that communities can recover within weeks to a few years of droughts, but increasing drought frequency and severity—including the spectre of multiyear events—mean that communities may fail to recover as quickly or completely after future droughts.

Drought can be transformative, tipping ecosystems to new states. Low flows interact with high water temperatures and nutrient enrichment to alter the outcomes of interactions between species. For example, filamentous algae may outcompete habitat-forming plants such as Ranunculus, profoundly altering ecosystem structure and function and potentially shifting systems to new stable states. Long-term time-series data require analysis to identify ecosystems approaching such shifts, to enable timely management action.

Manipulating just one thing—flow—can promote drought resilience. Management actions should seek to support natural drought-driven low flows—rather than drought-driven low flows exacerbated by abstraction, channel modification and other human impacts. In impacted rivers, physical habitat restoration as well as reducing abstraction can create more natural flow regimes, supporting species persistence and thus ecosystem functioning.

Manipulating other things—such as water quality—can promote resilience. Many environmental factors interact to determine responses to drought, each of which could be manipulated to promote resilience. For example, revegetating banks could reduce nutrient inputs and increase shade, reducing growth of nuisance species such as filamentous algae.

Refuges are key to recovery. Refuges—places in which drought impacts are lower—include isolated pools in largely dry reaches and fast-flowing habitats that persist in perennial reaches. High-quality, accessible refuges promote post-drought recovery, making their protection, creation and connection to impacted sites key management goals.

Ecosystems are everchanging. The dynamic nature of river ecosystems can be at odds with the notion of post-drought recovery as the return to a fixed pre-drought state. Instead, management actions should seek to restore natural ecosystem functioning, to support biodiverse communities within drought-resilient rivers that also deliver benefits to people.

2. Background

What is drought? Drought can be conceptualized from various perspectives, from meteorological to socioeconomic drought and including hydrological and ecological drought. All are defined by or result from a deficit in water, which is typically compared to a long-term average or other pre-defined threshold, and which causes a sufficiently "serious hydrological imbalance" (Seneviratne et al. 2012) to alter ecosystems and the services they deliver to people (Crausbay et al. 2020). Hydrological drought is specifically defined as a deficit in surface water and/or groundwater (Fleig et al. 2006), and manifests as abnormally low levels of streamflow in rivers and groundwater in their underlying aquifers (Van Loon 2015). Similarly, soil moisture drought indicates a water deficit within soil or, I suggest, unsaturated riverine sediments (Fleig et al. 2006). Ecological drought has long been ill-defined (Bachmair et al. 2016; Slette et al. 2019; IPCC 2022), but definitions of the term have recently emerged and emphasize the negative impacts of a deficit in water (potentially including rainfall, groundwater, streamflow and sediment moisture) on organism-to-ecosystem-scale structure and function (as summarized in Table 1) and thus on ecosystem services (Crausbay et al. 2017, 2020; also see Munson et al. 2021). Notably, Crausbay et al.'s (2017, 2020) conceptualization of ecological drought has been adopted by the US government (NIDIS 2023).

Drought is no longer a purely natural phenomenon. The above definitions typically conceptualize drought as natural, making drought part of the hydrological variability that supports biodiverse ecosystems (Everard 1996), with different species thriving in high-flow and low-flow periods (Bickerton 1995; Sarremejane et al. 2018). However, in rivers of the Anthropocene, drought interacts with other pressures, in particular water resource use (Crausbay et al. 2020; Klaus et al. 2022), and thus can no longer be conceptualized as a purely natural phenomenon (Van Loon et al. 2016).

Drought declarations relate poorly to ecological impacts. Environment Agency decisions to declare and end drought—which are based on a balance of information including river flows, groundwater levels, soil moisture levels as well as impacts on public water supply and the environment—are unlikely to represent the effects of drought on river ecosystems in either space or time. The often-gradual onset of ecological drought, its unclear end and its long-lasting impacts mean that it will routinely start before and end after an official drought. The Environment Agency Drought Monitoring Network—throughout which data are collected routinely, regardless of drought declarations—could enable better integration of ecological impacts into holistic drought monitoring systems (Hannaford et al. 2019).

Drought in river ecosystems. Drought alters the habitats within river ecosystems in many ways, reducing water depths and flow velocities in large rivers, causing rare dry phases in near-perennial reaches, and extending seasonal dry phases in winterbournes and other temporary streams. Drought disrupts all levels of biological organization, from the genes within a single organism to landscape-scale mosaics of aquatic and terrestrial habitats. Within river and stream ecosystems, all biological groups are affected, from microorganisms to fish—as well as species in riparian zones and wider terrestrial habitats. Taxonomic effects include changes in the abundance and distribution of individual species, potentially including local, regional and even national extinctions, although some species may thrive. These taxonomic changes have consequences for ecosystem functioning, for example altering the transfer of energy through food webs. And drought does not act alone in the Anthropocene; instead, it interacts with the many other human pressures impacting river ecosystems, including channel modification, pollution, and other climatic extremes including heatwaves.

Table 1. Direct responses of aquatic and terrestrial biotic groups to different types of drought: deficits in rainfall, groundwater and streamflow and sediment moisture. •, •• and ••• suggest the strength (i.e. 'importance') of the response, based on available evidence.

	Rainfall deficit	Groundwater deficit	Streamflow deficit	Sediment moisture deficit
Fishes	• A lack of rain may promote feeding e.g. by salmon parr (Stradmeyer and Thorpe 1987)	•• Reduced habitat quality and thermal refuges for coldwater species may reduce growth, survival and recruitment (Saltveit and Brabrand 2013)	••• Reduced submerged habitat restricts distribution, alters behaviour, causes mortality and lowers growth and recruitment (citations in main text)	• Increased mortality of fish with eggs or other life stages that tolerate unsaturated conditions e.g. European eel (Ellerby et al. 2001)
Aquatic macro-invertebrates	• Calm weather may promote water clarity and activity at the water surface (e.g. insect emergence)	•• Reduced availability of thermal refuges may reduce occurrence and abundance of coldwater taxa	••• Reduced diversity and abundance due to habitat loss (citations in main text)	•• Lower dry-phase moisture reduces survival of aquatic life stages (Stubbington and Datry 2013)
Terrestrial invertebrates (adults of insects with aquatic stages; terrestrial colonists)	• Calm weather may facilitate egg laying on water by adult insects (Everall et al. 2015), but increases abiotic harshness of dry in-channel	• Reduced upwelling water may increase availability of unsaturated streambed sediment habitats	•• An increase in diversity and abundance in drying habitats may subside after aquatic resources have	• Lower moisture in unsaturated streambed sediments reduces habitat quality

	habitats (Steward et al. 2022)	(Langhans and Tockner 2014)	been used up (Steward et al. 2022)	(Langhans and Tockner 2014)
Macrophytes	• Complex, positive and negative effects: dry weather typically promotes pollinator activity (Lawson and Rands 2019) but can reduce pollen germination and increase frost damage (Galen 2005)	• Altered water chemistry (e.g. increased nitrogen availability) may influence growth rates and patterns (Jansson et al. 2007)	••• Altered outcomes of competitive interactions between species can cause profound compositional change (citations in main text)	•• Lower dry-phase moisture reduces survival of desiccation-sensitive species (Hayes 2022)
Terrestrial and riparian plants		•• Reduced groundwater inputs can reduce taxa richness in riparian zones (Jansson et al. 2007)	•• Increased diversity and abundance due to increased habitat availability (Hayes 2022)	•• Lower dry-phase moisture reduces survival (Garssen et al. 2014)
Microbial biofilms	•• Reduced water supply may reduce survival during dry phases (Timoner et al. 2014)	• Altered water chemistry may influence microbial metabolism (Caramujo et al. 2008)	••• Reduced diversity and densities due to desiccation (citations in main text)	•• Reduced dry-phase moisture reduces survival and metabolism of surviving cells (Colls et al. 2019)

Unpredictability hampers drought research. Considerable research has documented ecological responses to drought in England's rivers. But—hampered by the unpredictable timing, typically gradual onset and sometimes unclear end, and widespread occurrence of drought, as well as the long timescales over which both drought impacts and recovery unfold—many studies (including those described herein) provide only local snapshots, focusing on impacts in the advanced stages of an event and/or initial recovery. In particular, studies characterizing patterns in a single drought year may be unable to disentangle the effects of drought from seasonal changes.

3. Scope of this review

I review the effects of drought (primarily deficits in surface water, and also including deficits in sediment moisture and influenced by low groundwater levels) on the biological communities in English rivers, encompassing microorganisms, plants, invertebrates and fish. I focus on benthic macroinvertebrates (hereafter, invertebrates), due to their abundance, biodiversity, ecological importance, ubiquitous occurrence and well-known environmental preferences—and thus their relatively well-documented responses to drought. In addition, invertebrates are central to river food webs, linking organisms at trophic levels from microorganisms to top predators, and thus enable exploration of whole-ecosystem responses to drought.

Although all riverine ecosystems (and in particular temporary streams, which shift between wet and dry in-channel conditions) support both aquatic and terrestrial species, the latter occurring both in channel margins and in adjacent riparian zones, I focus on aquatic species due to their reliance on rivers. I prioritize discussion of research done in English and other UK rivers, and consider all such river types in balance to their representation in the literature. I thus provide many examples from the considerable body of research documenting responses to drought in the chalk rivers and streams of southern England (Appendix 1). I nonetheless seek to represent other systems, including different natural river types and rivers exposed to different types and severity of human impacts—in particular, physical channel modification. I also consider relevant international literature, in particular where examples from England are lacking. Boulton (2003), Dewson et al. (2007), Boulton and Lake (2008) and Dollar et al. (2013) are among those providing broader reviews of the ecological effects of drought in river ecosystems.

Other reviews in this series consider the effects of drought on water quality (Bowes et al. 2023) and water temperature (Hannah et al. 2023). As such, the many and varied ecological responses to drought-driven changes in these environmental parameters are largely beyond the scope of this review, although I note a few key points.

A Glossary defines technical terms and general terms which may be open to interpretation.

4. How does drought alter habitats in river ecosystems?

Reflecting the five core components of the natural flow regime, droughts can be described by their frequency, magnitude, duration, timing and rate of change in flow conditions (Poff et al. 1997), with magnitude and duration collectively determining drought severity (Boulton 2003; Sarremejane et al. 2022), and with rate of change encompassing both drought onset and termination (Appendix 2). Drought can also be described by its spatial extent, which may be regional (i.e. sub-national), national or continental, although ecological impacts are typically felt at the smaller (habitat patch to catchment) spatial scales at which organisms interact with their environment. But how does drought manifest in a river channel? At any time, spatial variability can be considerable within a river network:

- In **perennial lower reaches**, drought effects may be reduced compared to upstream reaches, because reductions in flow velocity are less pronounced, sediments may already be silty and water depths may remain high enough to avoid exposing in-channel habitats (Wood and Petts 1994).
- In **perennial mid-reaches**, low flows include declines in depth that expose riffle crests, reductions in wetted width that expose marginal sediments, and/or decelerations in flow velocity that eliminate fast-flowing habitats (Stubbington et al. 2009a).
- In **near-perennial** and **previously perennial** reaches, rare and unprecedented dry phases, respectively, represent a profound, drought-driven change in in-channel habitat characteristics (Hill et al. 2019).
- In **seasonally intermittent** reaches, dry phases may start earlier and more abruptly, and dry-phase durations may become unusually long—sometimes just weeks longer, or entire winter wet phases can be missed (Bass et al. 2022).
- In **ephemeral** reaches, droughts may have limited effects because channels already have terrestrial habitat characteristics, although reduced soil moisture availability may alter plant community composition, with consequences for other biotic groups (Hayes 2022).

From low water depths in perennial reaches to dry phases in temporary reaches, no in-channel habitat state is unique to drought—but drought can change where in a network a particular state occurs (for example dry phases occur in near-perennial reaches only during drought) and extend the duration for which a state (e.g. discharge below a certain threshold) persists (Sarremejane et al. 2021).

The typical sequence of in-channel changes. At any one point in space, in-channel habitats can transition over time through the stages described above, starting with a reduction in depth which—in natural channels—reduces wetted widths, exposing marginal sediments and riffle crests (Figs 1–2; Boulton 2003). Decelerating flows deposit fine material including sediment on the bed (e.g. Wood and Petts 1994; Wright and Symes 1999), reducing habitat availability and diversity and clogging interstitial spaces (Vadher et al. 2015). Declining water volumes are less able to dilute solutes, potentially increasing concentrations of inorganic nutrients, organic matter, salts of both natural and anthropogenic origin, and many other pollutants—but changes can be unpredictable (Extence 1981; Wilby et al. 1998; Mosley 2015; Bowes et al. 2023).

Then, in temporary streams, water stops moving (i.e. flow cessation), still water habitats may contract to isolated pools, and ultimately most or all surface water may be lost, after which the water table may fall through the subsurface sediments and the moisture content of those sediments may decline (Figs 1–2; Boulton 2003). As such, the effects of drought on a specific in-channel location will change over months to years, generally increasing in magnitude to a maximum before the drought breaks, often with the rapid return of normal or even high flows (e.g. Kendon et al. 2013; Parry et al. 2016a).

Figure 1. In-channel conditions during drought in chalk streams: (a) extreme low flows; (b) a ponded reach; (c) an isolated pool and (d) a dry reach. Credits: Environment Agency.

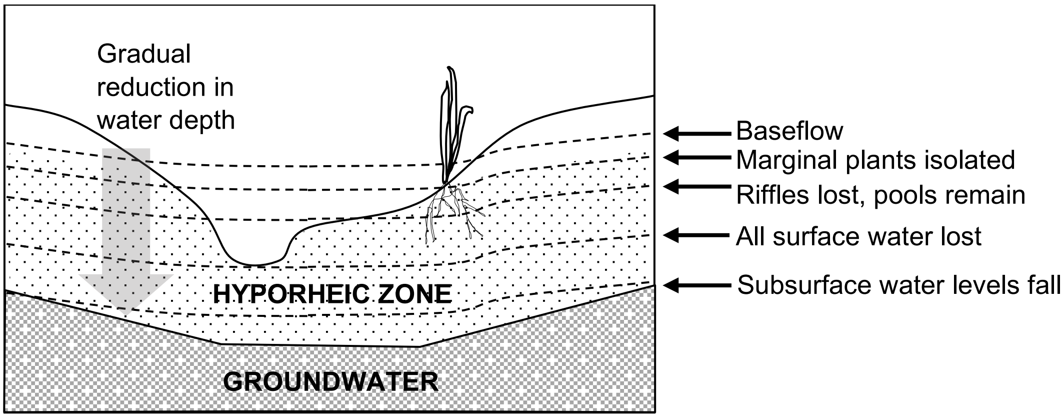


Figure 2. Stream cross-section, indicating key stages for aquatic biota during a gradual decline in discharge, as may occur during hydrological drought (adapted from Stubbington et al. 2009a).

Patterns vary between natural and modified river types. Drought unfolds differently depending on the nature and extent of human impacts on channel morphology, with natural channels proving more resilient (Holmes 1999; Dunbar et al. 2010a,b; also Calapez et al. 2014). Drought is also exacerbated by both surface water and groundwater abstraction: substantial increases in the duration of streamflow deficits have been observed across a wide range of river types (Wada et al. 2013; Van Loon et al. 2022), including chalk rivers affected by groundwater abstractions (Tijdeman et al. 2018, but see Hannaford et al. 2023).

Rain-driven flow increases can interrupt the typical sequence of in-channel changes. A meteorological drought (i.e. a deficit in precipitation) can be punctuated by unpredictable rainfall events of sufficient magnitude to cause streamflow to temporarily increase, in particular in surface-fed systems, with urban areas experiencing higher flow peaks due to limited infiltration of impervious surfaces (Hundecha and Bárdossy 2004). Climate change is expected to increase the occurrence of both summer storms and droughts in England; if so, the chances of storms occurring during droughts will also increase, potentially punctuating low-flow or dry periods with unpredictable high-flow events (Arnell et al. 2015).

Drought can increase water temperatures. In river reaches that retain surface water, low flows may coincide with hot spells and heatwaves in summer, resulting in increases in surface water temperature that reflect the reduced thermal inertia of smaller water volumes (Brooker et al. 1977; Hannah et al. 2023). Elevated temperatures are most likely in surface water-fed rivers and downstream reaches, whereas groundwater-dominated and/or shaded headwater streams are somewhat buffered against thermal variability (Berrie 1992; Wilby and Johnson 2020). High air temperatures can increase evaporation, reducing the size and persistence of pools and causing moisture loss from drying, damp and dry sediments (Gómez et al. 2017), in particular in unshaded reaches.

Is climate change affecting drought? To date, “no apparent trend in summer flows, low flows or drought” (Watts et al. 2015) has been identified in English rivers (also see Hannaford and Buys 2012; Hannaford 2015; Hannaford et al. 2023). However, climate change may already be increasing drought frequency, magnitude, duration, severity and spatial extent in England, and may also be altering drought characteristics including its timing and rate of onset and termination (Spinoni et al. 2015; Hall et al. 2022). Streams once considered perennial are drying for the first time, in England (Hill et al. 2019) as in mainland Europe (Crabot et al. 2021). Moreover, there is greater confidence in predicted future increases in UK drought frequency, magnitude, duration, severity and spatial extent (Spinoni et al. 2018; Rudd et al. 2019; Hannaford et al. 2022), and the risk of flash droughts—defined by their rapid onset and intensification—is increasing (Pendergrass et al. 2020; Walker and Van Loon 2023), including in humid regions of Europe (Yuan et al. 2023).

Given its often-gradual, inherently unpredictable, hard-to-define onset, any change in drought timing is uncertain.

Ecological responses to drought

In sections 5–9, I explore ecological responses to different drought-driven in-channel conditions, including low flows and—in temporary streams—flow cessation, pool formation and dry phases. In many English rivers, it is likely that manifestation of these conditions will be influenced by concurrent abstraction pressures, as well as wider river regulation.

At the end of each section, I suggest the ecological effects of the predicted increases in drought occurrence—including its frequency, magnitude, duration, severity and/or spatial extent—as well as changes to drought timing and rate of onset and termination. These suggestions are based on limited evidence, given the few studies that have sufficient time series to investigate the effects of these drought characteristics (but see Sarremejane et al. 2019, 2020, 2021).

5. Low flows alter densities, abundance, biomass and richness

Moderate low flows can increase the abundance and biomass of competitive species. Where water depth, wetted width and flow velocities remain sufficient, low flows can represent periods of environmental stability that enable competitive taxa with broad resource and habitat requirements to become increasingly abundant. Low flows may also increase food availability for invertebrates and fish:

- In spring and summer, light penetration of shallow water promotes algal growth on sediment and macrophyte substrates, and low flow velocities minimize disruption and detachment of thickening, diatom-rich biofilms. High biofilm biomass supports abundant populations of ‘grazer’ invertebrates including Baetidae mayflies and Chironomidae midges (Wright et al. 2003; Romaní et al. 2013), and can increase their growth and fecundity, as reported for the blue-winged olive mayfly Serratella ignita (Wright and Symes 1999). Aquatic and terrestrial plant growth can also allow herbivores such as snails to increase in abundance during drought (Wood and Petts 1994). Such increases may be short-lived, depending on how drought conditions subsequently unfold; in particular, biofilm-coated macrophytes experience notable declines in growth and abundance (see section 6).

- In autumn, low flows allow leaf litter to accumulate, supporting ‘shredders’ (which consume coarse particulate organic matter) such as Gammarus shrimps (Argerich et al. 2008).

If submerged habitats shrink, animal densities increase. Depending on channel shape, declining discharge can cause submerged habitats to contract, forcing motile organisms including fish and many invertebrates to move into, and share, a shrinking habitat area (Armstrong et al. 1998; Pařil et al. 2019). As a result, densities (i.e. the number of organisms per unit area) but not necessarily abundance (i.e. the number of organisms present) of motile species such as Gammarus shrimps, Chironomidae midges (Wright 1992; Wood and Petts 1999; Stubbington et al. 2011; Aspin et al. 2019a), brown trout Salmo trutta and Atlantic salmon Salmo trutta (Stradmeyer et al. 2008) can temporarily increase. For example, Verdonschot et al. (2015) recorded peak invertebrate densities and richness after pools became isolated in a sandy lowland stream. These increased population densities can cause biotic interactions to intensify; for example Stradmeyer et al. (2008) recorded increased aggression by high-ranking, dominant brown trout and reduced movement and feeding by other fish after pool formation.

In contrast, densities of immotile species remain stable—and organisms stranded in drying marginal or mid-reach sediments may die, including species of conservation concern such as the freshwater pearl mussel Margaritifera margaritifera (see section 11; Cosgrove et al. 2021; Nogueira et al. 2021).

If submerged habitats shrink, abundance decreases. Although densities can increase as discharge declines, abundance and biomass can decrease, in particular in channels with naturally variable bed forms, due to the gradual reduction in submerged habitat availability and diversity (Wood and Petts 1994, 1999). Cowx et al. (1984) measured invertebrate abundance in a 1-m transect after declines in wetted width and thus submerged habitat availability in a small upland stream during the 1976 drought: abundance was 60% lower than in 1977, a non-drought year. Accordingly, despite their increasingly high densities as submerged habitats contract within a single drought year (Stubbington et al. 2011), the abundance of the common, often-dominant shrimp Gammarus pulex/fossarum, for example, is higher in years with higher discharge (Stubbington et al. 2009a). Low discharge and slow flow velocities can also initiate downstream drift of invertebrates including Baetidae mayflies and Simuliidae blackflies (Corrarino and Brusven 1983; Brittain and Eikeland 1988), reducing their local abundance.

Abiotic and biotic factors interact to determine knock-on effects of higher densities. All things being equal, the combination of declining organism abundance (i.e. diminishing food resources for predators) and increasing densities (i.e. rich pickings from a smorgasbord of prey) can increase biotic interactions including competition and predation, thus reducing the abundance of poor competitors and prey species (Wright and Symes 1999), in particular where wetted habitats decrease in extent (Aspin et al. 2019a). However, all things may not be equal: drought may independently cause concurrent declines in the abundance of predatory invertebrates (Ledger et al. 2013) and fish (Cowx et al. 1984), thus reducing pressure on prey species.

Drought-driven low flows alter taxonomic richness. If decreasing discharge forces motile organisms (including benthic invertebrates) to share a diminishing submerged habitat area, taxonomic richness can temporarily peak, with the co-occurrence of flowing-water and still-water taxa leading to unusual assemblages in low-flow impacted reaches (Extence 1981; Perrow et al. 2007; Verdonschot et al. 2015). Over time, as habitat availability, quality and diversity decline, local-scale richness can gradually decrease (Wood et al. 2010; Stubbington et al. 2015; Aspin et al. 2019a). Such decreases likely reflect a combination of stochastic and deterministic local extinctions of tolerant and sensitive taxa, respectively, driven by chance events such as predation and environmental determinants such as water quality, respectively (Sarremejane et al. 2021a).

However, at the river and catchment scales, few taxa are lost—most persist in some reaches (Wood and Petts 1999; Wright et al. 2003; Sarremejane et al. 2020). Based on evidence from perennial sites, average taxonomic richness can therefore be comparable in drought and non-drought years (Wright and Symes 1999; Wright et al. 2002). In contrast, in rivers dominated by perennial flow but also including near-perennial and seasonally intermittent reaches, river-scale richness can be lower in years in which ponded and/or dry phases last longer in time (Sarremejane et al. 2020).

If climate change increases the occurrence of drought-driven low flows:

- Any increase in low flows could increase the dominance of competitive generalist species at the expense of overall community biodiversity.
- Any increase in drought magnitude could reduce submerged habitat availability, causing the abundance of aquatic species to decline.
- Any increase in the river-scale spatial extent of low flows could limit the availability of refuges (e.g. fast-flowing or deep waters for species requiring such habitats), reducing metacommunity health and limiting post-drought recovery.

Abundance, biomass, densities and richness provide limited insight into community responses to drought. Total abundance, densities, biomass and the average mixed-level taxonomic richness of a community are among the metrics most commonly used to summarize community responses to drought, but should be treated with caution. Regardless of whether they increase, decrease or remain stable, none of these metrics represent community composition (i.e. the taxa responsible for observed values) and they can thus overlook profound compositional change including taxon losses and gains (i.e. taxonomic turnover). These issues highlight the importance of concurrent consideration of community composition alongside summary metrics, as discussed in section 6.

In particular, taxonomic richness can remain stable as drought-driven flow declines replace riffles with still-water habitats, if the loss of flow-loving taxa is offset by the arrival of still-water taxa (Hill and Milner 2018). Similarly, Aspin et al. (2018) observed considerable taxonomic turnover along a gradient of increasing drought intensity in an experimental stream, reflecting the loss of drought-sensitive taxa and the arrival of tolerant taxa. In addition, the capacity of richness to summarize taxonomic responses to drought is routinely limited by the identification of many common taxa to a resolution coarser than species. For example, identifying the ubiquitous true fly family Chironomidae to family, subfamily or tribe overlooks species-specific differences in drought tolerance (Cañedo-Argüelles et al. 2016).

At the population (i.e. single species) level, metrics such as abundance can fail to detect changes in population structure, and thus fail to recognize the risk of future population collapse if all individuals are non-reproductive adults (Cowx et al. 1984; Pařil et al. 2019). Determination of population structure—although rarely done for groups other than fish—is necessary to understand the long-term effects of drought, for example the effects of drought on physiology and, in particular, reproductive fitness.

At the site scale, changes in abundance and richness can vary considerably among habitat patches (Wright et al. 2003); at the river or network scale, declines at some sites can be compensated by gains at other sites (Wright et al. 2004; Sarremejane et al. 2020). As such, characterization of responses to drought partly depends on the spatial scale(s) at which patterns are studied.

6. Low flows alter habitats, resources and community composition

Low flow velocities reduce the occurrence of flow-loving taxa. Many taxa are eliminated as flow declines due to the loss of their preferred habitats, such as gravel-dominated, fast-flowing riffles (Wright et al. 2002; Chadd et al. 2017). Flow-loving invertebrate families whose occurrence declines during low flows include Baetidae mayflies, Elmidae beetles, Leuctridae stoneflies, Piscicolidae leeches, Hydropsychidae and Rhyacophilidae caddisflies, and Simuliidae blackflies (Wright and Berrie 1987; Wright and Symes 1999; Wood and Petts 1994; Wright et al. 2004; Ledger et al. 2012). In addition, many flying adult insects lay eggs only on fast-flowing water, which can reduce the occurrence of aquatic juveniles the year after a drought (Iversen et al. 1978; Cowx et al. 1984).

Low flow velocities increase the occurrence of still-water taxa. Whilst flow-loving taxa decline, those associated with still waters (which develop in the margins of perennial reaches and may become extensive in temporary reaches) can become more common, including Dytiscidae and Haliplidae beetles and Notonectidae true bugs (Wood and Petts 1994; Wright et al. 2004; Perrow et al. 2007). As such, low flows can increase riverine habitat availability for still-water taxa (Perrow et al. 2007; Hill and Milner 2018), potentially compensating for drought-driven declines in wetland habitats in the wider catchment. Semi-aquatic taxa including many true fly families (e.g. species of Ceratopogonidae, Psychodidae and Stratiomyidae) can also become more common as drought progresses (Ledger et al. 2012), with their aquatic juveniles developing rapidly after egg deposition by flying adults (Aspin et al. 2018). These still-water beetles, true bugs and true flies could, in theory, include species of conservation concern e.g. the Nationally Rare yellow-tipped soldier fly Oxycera terminata (Armitage and Bass 2013; Drake 2017; Aspin et al. 2018) and the diving beetle Rhantus suturalis (Perrow et al. 2007; Foster 2010).

Low flows reduce depths, increase temperatures, reduce oxygen—and kill animals. Declining water depths increase the influence of solar radiation on water temperatures, in particular during summer hot spells and longer heatwaves. Moderate increases in temperature can accelerate life cycle completion (Everall et al. 2015), potentially leading to trophic mismatches—i.e. the disruption of links between consumers and their food resources (Larsen et al. 2016).

Rising temperatures inflict physiological stress on coldwater and then also species with broader temperature tolerances (Williams 2016). Coldwater invertebrates include many mayflies, stoneflies, caddisflies (Macadam et al. 2022) and flatworms (Reynoldson 1953; Durance and Ormerod 2010). Coldwater fish include the salmonids Atlantic salmon Salmo salar, brown trout Salmo trutta and grayling Thymallus thymallus, as well as bullhead Cottus gobio, brook lamprey Lampetra planeri, river lamprey L. fluviatilis and stone loach Barbatula barbatula (Buisson et al. 2008, 2013; Lassalle and Rochard 2009; Moss 2014). The occurrence of thermally sensitive species is greater in upland than lowland freshwaters (Macadam et al. 2022), perhaps because temperatures at low altitudes are already too high for some species (e.g. Kitchen et al. 2010). Within a river network, the communities in cooler shaded and/or groundwater-dominated headwater streams may include species that are particularly vulnerable to higher temperatures (Biondi et al. 2022; Macadam et al. 2022).

The physiological stress associated with increasing temperatures can be direct or indirect due to associated reductions in dissolved oxygen availability. This stress can result in behavioural change, increased disease susceptibility, reduced reproductive fitness and ultimately increased mortality of individuals and thus population declines (Wood et al. 2001; Jonsson and Jonsson 2009; Wood et al. 2010; Pařil et al. 2019) or even local extinctions. Fish kills during low flows are particularly evident and alarming. For example, mass mortality of adult salmon in the River Wye during the 1976 drought was linked to low flows and warm temperatures, which first promoted growth of the submerged macrophyte water crowfoot (Ranunculus species) but then caused the plant’s death and decay, causing severe deoxygenation of the water (Brooker et al. 1977). Similarly, Cowx et al. (1984) attributed elimination of an entire brown trout age class from a small upland stream to the physiological stress caused by high water temperatures, which peaked at 26.5°C during the period of lowest flows. More recently, 10 fish kills were classified as major incidents attributed to dry weather affecting England in summer 2018 (Turner et al. 2021).

Low flow velocities deposit fine sediment on the streambed—and kill animals. Slow flow velocities promote deposition of silt and sand on the streambed, and this fine sediment can accumulate and persist, in particular in low-energy systems such as chalk streams (Wright and Berrie 1987; Wood and Petts 1994, 1999; Ledger and Hildrew 2001). Fine sediment has a wide range of detrimental, sometimes lethal, effects on biotic groups from biofilms to fish (Wood and Armitage 1997; Kemp et al. 2011; Jones et al. 2012a,b).

Compared to gravel, Ranunculus and other macrophytes, silty sediments support lower densities of most invertebrate taxa, including Simuliidae blackfly larvae, which need firm substrates to attach to; and Glossosomatidae caddisfly larvae (Wright and Berrie 1987) and Baetidae mayflies (Wright et al. 1981), which need suitable substrates from which to graze biofilms. For salmonids, fine sediment reduces the quality of spawning habitats (Wood and Armitage 1997). In contrast, densities of a few invertebrate taxa are higher in silt than other habitats, including Sphaeriidae clams and Chironomidae midge larvae, which feed on the accumulating detritus (Wright and Berrie 1987; Wright 1992; Ledger et al. 2012). Their predators, such as Sialidae alderflies (Wood and Petts 1994) and the burrowing green drake mayfly Ephemera danica (Everall et al. 2015), may thus also increase. A few fish also favour substrates dominated by fine sediments (and organic particles), including spined loach Cobitis taenia (Robotham 1978).

Slow flows and siltation alter habitat-forming plant communities. Flow is one of the key influences on riverine macrophytes including species of the flow-loving genus Ranunculus, which forms a fundamental component of habitat structure in systems including chalk streams (Wilby et al. 1998; Gurnell et al. 2006). As flow velocities fall, silt is deposited on the streambed and colonized by emergent vegetation (Fig. 3a), which shades and outcompetes submerged macrophytes including Ranunculus (Wade et al. 2002). At the same time, the lack of scouring flows—in particular during summer (Wilby et al. 1998; Franklin et al. 2008)—allows algae to accumulate on macrophyte leaves and stems, reducing their capacity to photosynthesize (Fig. 3b). As a result, established stands of plants including Ranunculus decline in health and abundance (Ladle and Bass, 1981; Cranston and Darby 2002; Westwood et al. 2006a,b), displacing the diverse range of invertebrates they support (Wright and Berrie 1987; Wright et al. 1998; Wright 1999), with consequences that extend throughout riverine food webs (McIntosh 2019). Moreover, Ranunculus and other flow-loving plants may not establish populations in drought years if flows fail to meet their growing season requirements (Wright 1999).

Biofilms, macroalgae and phytoplankton can thrive in low-flow conditions, especially in warm, nutrient-rich waters, such as occur during summer in lowlands dominated by agricultural land (Fig. 3; Wilby et al. 1998; Wright et al. 2004; Bowes et al. 2016). Although such algal growth can benefit certain invertebrate species, algae provide a simpler habitat and support lower invertebrate biodiversity than the Ranunculus they often replace (Wright et al. 2004). Moreover, both night-time respiration by living algae and decomposition of dead algae by heterotrophic microorganisms severely reduce dissolved oxygen concentrations during drought (Parr and Mason 2003; Bowes et al. 2023), causing high-profile fish kills as well as invertebrate mortality (Brooker et al. 1977; Parr and Mason 2003).

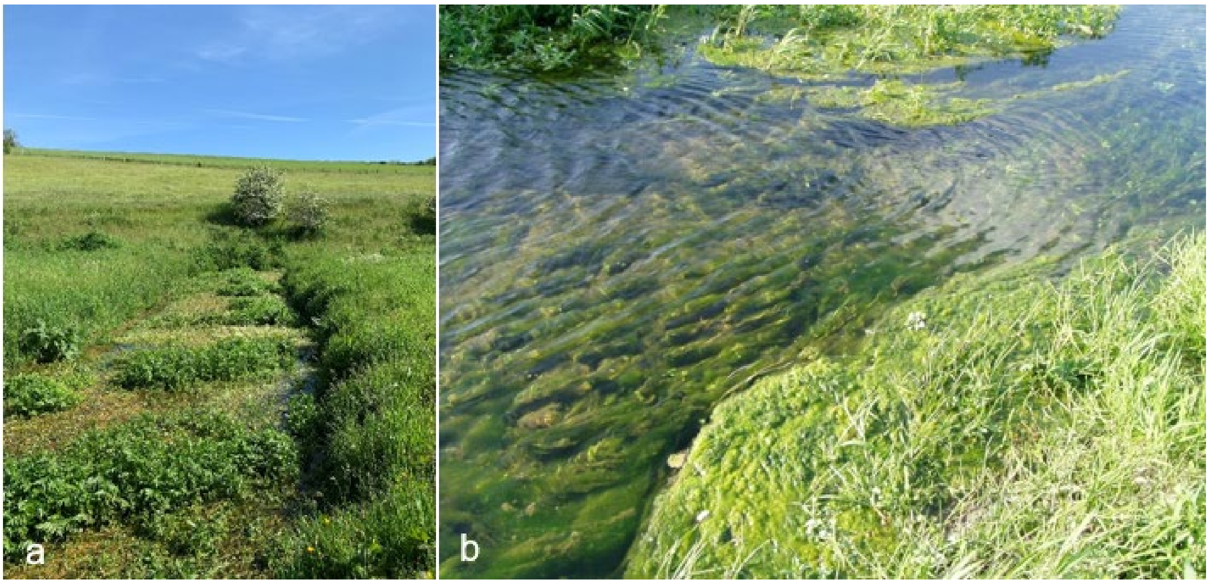


Figure 3. (a) Emergent plants thrive in a ponded stream; (b) Algae covering submerged mid-channel Ranunculus and accumulating around emergent marginal plants as flows recede during drought in a chalk stream. Credit: (a) T. Sykes; (b) N. Holmes.

Drought-driven losses of sensitive taxa homogenize network-scale communities. At river-to-network spatial scales, drought-driven reductions in habitat diversity increase community similarity among sites by eliminating different, infrequently occurring drought-sensitive taxa from individual sites and reducing the remaining community to a core of common, tolerant taxa (Ledger et al. 2012; Stubbington et al. 2015). Equally, communities can sometimes share fewer and fewer taxa as a drought unfolds, due to stochastic extinctions as well as spatial variability in the taxa persisting in the remaining slow-flowing, shallow and/or ponded habitats (Aspin et al. 2018; Sarremejane et al. 2018).

Low flows can increase salinity in estuarine reaches. Drought-driven reductions in water volumes can reduce inputs of fresh water into estuarine reaches (Attrill et al. 1996) altering their water chemistry, including an increase in salinity (Attrill and Power 2000a). Associated biotic changes include the loss of saline-sensitive invertebrate taxa such as Caenidae mayflies and the Nationally Scarce depressed river mussel Pseudanodonta complanata (NBN 2021). Drought-related changes in the population densities of dominant estuarine taxa including caridean and gammarid shrimps also occur, driven primarily by temperature, altering the structure and function of estuarine food webs (Attrill and Power 2000b).

If climate change increases the occurrence of drought-driven low flows:

- The risk of elevated water temperatures (and associated declines in dissolved oxygen) eliminating thermally sensitive species, including salmonids, will increase. Such declines in water quality are likely to be most severe in drier parts of south-east England (Bowes et al. 2023).
- The growth and survival of habitat-forming macrophytes including Ranunculus will decrease, altering ecosystem structure and function and reducing biodiversity.

- The risk of stochastic extinctions could increase—and whilst many such losses have little impact on ecosystem function, a few could have unexpected, far-reaching consequences.

7. Dry phases: persistence of pools and other wet refuges

The reproductive fitness of pool refugees may decline. As described in section 5, as flows recede during drought, fish and many invertebrates become confined to shrinking submerged habitats. When these wet habitats become disconnected and form isolated pools, organisms may prosper or perish (Elliott 2000). Persistent pools can act as refuges for small fish including bullhead Cottus gobio and the sticklebacks Gasterosteus aculeatus and Pungitius pungitius (Perrow et al. 2007), as well as brown trout Salmo trutta (Morrison 1990). However, the reproductive potential of pool survivors has very rarely been studied, and may be low. For example, Pařil et al. (2019) provide a rare study of a Gammarus shrimp population in a small Czech stream: drought caused water to recede to a few isolated pools during two 10–20-day periods in one summer, causing a substantial population decline that included loss of most spring-recruited juveniles and all reproductive females. For fish, stocking by river managers can influence post-drought community composition (Perrow et al. 2007).

Figure 4. A dead brown trout Salmo trutta in a small, isolated pool in the bed of the Bourne Rivulet, a chalk stream. Credit: N. Howard-Jones.

Pool refuges may become traps in which organisms die. Although wet refuges as small as a "puddle of water" can enable 'sensitive' taxa including caddisflies, crustaceans and mayflies to survive in an otherwise dry reach (Perrow et al. 2007), persistent pools vary in quality. Some can become traps in which invertebrates and fish (in particular larger fish) die if water quality becomes too poor (Fig. 4; Elliott 2000; Vander Vorste et al. 2020). In particular, pool water quality is typically characterized by high temperatures and pollutant concentrations and low oxygen concentrations (Fausch et al. 2002; Ruiz-Navarro et al. 2016), and oxygen availability may fall below the levels required by many fish, in particular as water temperatures rise during hot spells and heatwaves (Cowx et al. 1984; Elliott 2000). Similarly, within a week of a peak in invertebrate richness and densities associated with pool isolation in a sandy lowland stream (see section 5), richness decreased sharply in association with declining oxygen concentrations (Verdonschot et al. 2015).

If climate change alters conditions in dry-phase pools:

- If increases in water temperatures and associated reductions in dissolved oxygen levels become more severe, the survival of animals in pool refuges will decline.
- If drought magnitude, duration and severity increase, pools will become less likely to persist, instead becoming traps in which organisms die.

8. Dry phases: complete surface water loss

Pool drying kills aquatic organisms. If pools dry, “catastrophic mortality of fishes” can occur (Archdeacon and Reale 2020), and other desiccation-sensitive aquatic organisms including invertebrates are also likely to die (Matthews and Marsh-Matthews 2003; Stubbington et al. 2017). Even species whose life cycles are timed to coincide with seasonal shifts between flowing and dry phases in temporary streams (i.e. brown trout Salmo trutta and bullhead Cottus gobio) may be scuppered by rare, drought-driven dry phase in a near-perennial or previously perennial reach (Wright and Berrie 1987), or the early onset of a dry phase in a seasonally intermittent stream. For example, a rare dry phase eliminated brown trout from a small upland stream (Hynes 1958).

The subsurface sediments of the hyporheic zone can be an extensive refuge. Despite widespread mortality as rivers dry, local dry-phase refuges (i.e. any habitat retaining moisture or water) provide a glimmer of hope for stranded organisms, in particular the saturated subsurface sediments of the hyporheic zone (Chester and Robson 2011; Stubbington 2012). Depending on the composition of the streambed sediments, a few fish, including eel Anguilla anguilla (Tesch 2003), juvenile lamprey (Lampetra species; Rodríguez-Lozano et al. 2019), Atlantic salmon Salmo salar parr (Debowski and Beall 1995) and loach (Kawanishi et al. 2013) can burrow into the subsurface sediments, and this hyporheic zone is also an important dry-phase refuge for many invertebrates (Stubbington 2012; Vander Vorste et al. 2016a).

The hyporheic refuge varies in quality. The hyporheic zone’s capacity to provide a long-term refuge in which organisms survive a dry phase depends on interactions between its habitat characteristics and species-specific environmental tolerances and traits (Stubbington 2012; Loskotová et al. 2019). Then, if water is lost or if (as is likely) oxygen concentrations fall low, the hyporheic zone can become a graveyard in which desiccation-sensitive organisms die (e.g. Young et al. 2011)—although a wide range of aquatic invertebrates persist as desiccation-tolerant forms (Stubbington and Datry 2013).

Moreover, clogging of interstices by fine sediment can prevent most animals from accessing subsurface sediments in the first place (Vadher et al. 2015), and some streams (e.g. bedrock-dominated channels) lack any hyporheic zone.

Other refuges. In addition to the hyporheic zone, even localized, moist (but not wet) refuges such as thick, dried algal mats may reduce evaporation and provide dry-phase refuges for invertebrates, promoting community resilience (Ledger and Hildrew 2001; Nelson et al. 2021). However, the metabolic health of aquatic organisms declines in the absence of free water, and the reproductive potential of such refugees is unknown (but see Pařil et al. 2019).

If climate change increases the occurrence of drought-driven dry phases:

- The survival of fish, invertebrates and other species in dry reaches will increasingly depend on the availability, accessibility and quality of dry-phase refuges including the hyporheic zone.
- If drought magnitude, duration and severity increase, dry-phase refuges including the hyporheic zone will decline in quality, reducing survival of even some desiccation-tolerant organisms.

9. Rare drought-driven dry phases cause biodiversity loss

Few aquatic species are unaffected by rare dry phases (in near-perennial reaches) and unprecedented dry phases (in previously perennial reaches). Most taxa occur at lower densities after flow returns, a few at comparable or higher densities, and some are lost (or remain undetected during a study period; Table 2 e.g. Hynes 1958; Ladle and Bass 1981). Reported densities vary depending on when organisms are sampled, but the duration between flow resumption and sample collection is not always known. As such, the patterns below describe populations sampled weeks to months (but <1 year) after flow resumption.

- Densities of some common, dominant taxa such as Gammarus shrimps, which support energy transfer through food webs through their ‘shredding’ of leaf litter, can be much **lower** after a dry phase (Ladle and Bass 1981; Wood and Petts 1999; Perrow et al. 2007; Ledger et al. 2011).
- Densities of desiccation-tolerant taxa of oligochaete worms and true fly larvae as well as some Limnephilidae caddisflies (Ladle and Bass 1981) can be **comparable** before and after a dry phase. Hynes (1958) also recorded invertebrates including flatworms “as soon … as the water table rose” after a 10-week dry phase in a small mountain stream.
- Densities of taxa including the blue-winged olive mayfly Serratella ignita and Lymnaeidae snails can be **higher** the year after flow resumes (Ladle and Bass 1981); they may benefit from available habitat space as well as reduced predation by and competition with usually dominant taxa such as Gammarus shrimps (Kelly et al. 2002).

- **Losses** (i.e. taxa that are not detected post-drought during a study) include many stoneflies, the caddisfly genera Agapetus (Glossosomatidae), Silo (Goeridae) and Sericostoma (Sericostomatidae), and Piscicolidae leeches. Species-level identification also reveals losses within major taxa often considered drying tolerant, such as Nais oligochaetes (Ladle and Bass 1981; Aspin et al. 2018).

If climate change increases the occurrence of drought:

- Differences in drying sensitivity will alter the outcomes of competitive interactions among species, shifting communities towards greater dominance of tolerant taxa.
- The risk of local and regional species losses will increase, with desiccation-tolerant species filling the ecological niches left by lost taxa, reducing overall biodiversity.

10. Drought alters communities in naturally temporary streams

In temporary streams, drought-driven changes to wet-phase conditions and associated ecological responses are likely to be largely comparable to those in perennial streams, as considered above. I therefore focus here on the effects of drought-driven changes in dry-phase characteristics. This section is informed largely by evidence from winterbourne chalk streams, as England's best-studied type of temporary stream.

Communities are adapted to seasonal shifts between dry and wet phases. Most species inhabiting winterbournes and other seasonally intermittent streams have traits enabling their survival during a dry phase and/or their capacity to recolonize after water returns, including microorganisms (Romaní et al. 2017), invertebrates (Stubbington et al. 2017), plants (Sabater et al. 2017) and fish (Kerezsy et al. 2017). Winterbourne insects include specialists such as the scarce purple dun mayfly Paraleptophlebia werneri and the winterbourne stonefly Nemoura lacustris, which have lifecycle adaptations in which the timing of terrestrial and aquatic life stages coincides with dry and wet phases, respectively (Fig. 5; Macadam et al. 2021). Similarly, brown trout Salmo trutta and bullhead Cottus gobio have behavioural strategies that enable their use of high-quality habitats in winterbournes during flowing phases then their migration downstream into perennial reaches before a dry phase begins (Solomon and Templeton 1976; House et al. 2008). These lifecycle adaptations promote both escape as flow declines and recolonization after flow returns.

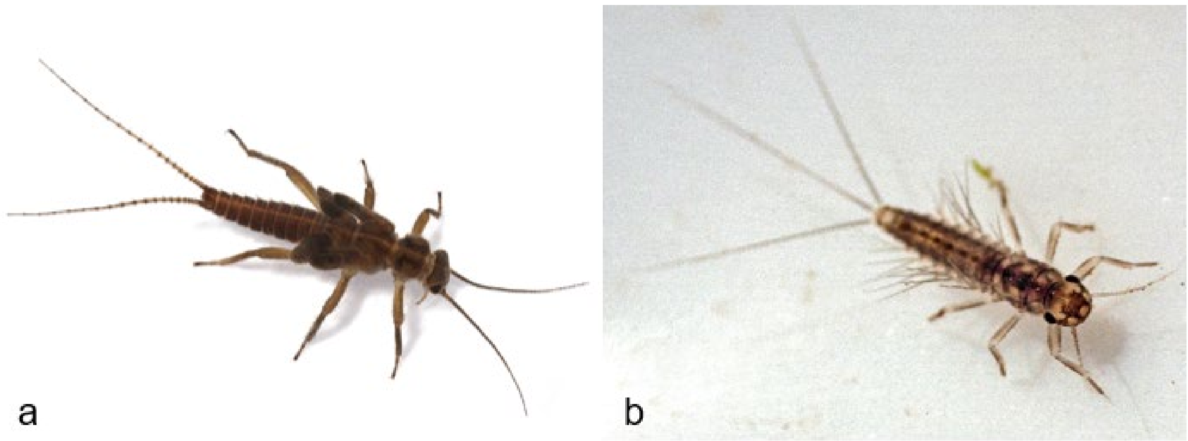


Figure 5. The aquatic juvenile life stages of the winterbourne specialist insects (a) the winterbourne stonefly Nemoura lacustris and (b) the scarce purple dun Paraleptophlebia werneri. Credits: (a) Cyril Bennett; (b) Adrian Chalkley.

Drought alters the seasonality of dry and wet phases. In temporary streams, drought-driven changes to flow regime characteristics include unusually early and/or rapid-onset drying and unusually long dry phases. These conditions can overcome species' adaptations, reducing population densities and biodiversity and altering community composition, as detailed below:

– **Longer dry-phase durations, shorter wet-phase durations.** Some terrestrial adult insects lay desiccation-tolerant eggs on dry sediments, which then develop once water returns (e.g. the winterbourne stonefly; Tapia et al. 2018). During drought, longer dry-phase durations could increase the sediment moisture deficits that result from rainfall deficits, with drier sediments potentially surpassing the desiccation levels that eggs can tolerate (Stubbington and Datry 2013). Any increase in dry-phase duration increases the likelihood of mid-drought hot spell or heatwave occurring, increasing dry-phase magnitude, as described below.

– Unusually long dry phases equate to unusually short wet phases that start later and/or end earlier (Bass et al. 2022), potentially reducing the proportion of a cohort of aquatic juveniles that develops and emerges as terrestrial adults in time to avoid desiccation during the next dry phase (Drummond et al. 2015).

– **Higher dry-phase magnitude**. Summer droughts can equate to below-average rain falling on dry streambeds. Along with any increase in dry-phase duration, an associated reduction in sediment moisture content could push the persisting egg, juvenile and adult life stages past their desiccation tolerance limits (Welton et al. 1987; Stubbington and Datry 2013). Such moisture loss is particularly likely in channels in which evapotranspiration is exacerbated by a lack of riparian shading and associated in-channel plant growth (for example in agricultural areas; Hayes 2022) and during hot spells and longer heatwaves.

– **Changes to dry-phase frequencies**. In theory, 'false starts' (i.e. short wet phases which precede redrying and a subsequent 'true', long-term flow resumption) could trigger development of desiccation-tolerant insect eggs into desiccation-sensitive juveniles that die during subsequent, short dry phases. However, evidence for this hypothesized pattern is lacking (Strachan et al. 2016; Stubbington et al. 2016). Instead, bet-hedging strategies such as asynchronous egg hatching can enable populations to tolerate some natural, interannual hydrological variability (Lytle and Poff 2004).

- **Early drying and late wetting**. The timing of drought-driven changes in habitat conditions—in particular, dry and wet phases—determines effects on species that have season/month-specific life cycle activities such as egg laying, egg hatching, migration and adult emergence. Current evidence, described below, suggest that insects, fish and other organisms in temporary streams can cope with interannual variability in hydrological conditions and may take drought-driven changes in their stride… for now.

    - Dry phases may start earlier in a drought year, which—alongside rising temperatures—could affect species with seasonal lifecycle activities (Parmesan 2006). For example, the proportion or size of juvenile aquatic insects that emerge as adults before water is lost may decline if rivers dry sooner (Tapia et al. 2018). However, bet-hedging strategies (Lytle and Poff 2004), in this case asynchronous juvenile development and adult emergence, can promote population persistence (Robson et al. 2011). In addition, phenological plasticity may allow insects to emerge earlier in dry years (Leberfinger et al. 2010)—although doing so can cause trophic mismatches, disrupting links between organisms and their seasonal food resources (Larsen et al. 2016). Early dry-phase onset could also increase the proportion of fish that do not migrate downstream in time to escape drying. However, whilst some brown trout become stranded in dry-phase refuges, declining flows trigger others to migrate into downstream perennial reaches (Wright and Berrie 1987; House et al. 2008).

    - The terrestrial adults of some insects lay their eggs on water, and if water returns later than normal during drought, this could prevent egg laying, as reported for the caddisflies Goera pilosa, Hydropsyche angustipennis and Rhyacophila obliterata and the mayfly Ephemerella ignita (Hynes 1958; Iversen et al. 1978). Late wetting and flow resumption could also delay upstream fish migration or prevent access to spawning grounds, with potential consequences for seasonal life stage completion (Wright and Berrie 1987).

- **Rapid drying.** Winterbournes and other temporary streams typically dry slowly (Moon 1956), giving organisms time to react—physiologically and/or behaviourally—for example by slowing their metabolism and entering a desiccation-tolerant dormant state (Strachan et al. 2015), by burrowing into saturated or moist streambed sediments (Poznańska et al. 2015; Patel et al. 2021), or by swimming or drifting downstream (House et al. 2008). If drought increases the rate of wet-to-dry transitions (for example during flash droughts; Pendergrass et al. 2020; Yuan et al. 2023), this could shorten the time available to start and complete these activities, as observed for migrating coho salmon Oncorhynchus kisutch in US streams (Kastl et al. 2022). In addition, faster wet-to-dry transitions could shorten the period between the environmental cues that trigger insect metamorphosis and its completion, reducing the proportion of individuals that emerge as adults (Drummond et al. 2015). However, the extent of water loss (e.g. if pools persist; how dry sediments become) is likely to be a greater influence on aquatic species than how water is lost.

– **Rapid wetting.** Even during drought (or marking the point of drought termination e.g. Kendon et al. 2013; Parry et al. 2016b), if a dry phase is interrupted or ended by a flood or other high-flow event, organisms can be washed from refuges including the subsurface sediments of the hyporheic zone, especially if sediments are mobilized (Stubbington et al. 2009b). Any additional floods during the period after flow resumes could also compromise community recovery.

If climate change alters dry-phase and wet-phase characteristics:

- Future climate-change-driven shifts in drought severity and timing may soon go beyond the range of conditions that even adapted species can tolerate, reducing biodiversity in naturally temporary streams.
- Any increase in the occurrence of floods during drought termination phases could interrupt community recovery after flow resumes, reducing biodiversity in temporary streams.

11. Drought threatens species of conservation concern

As discussed below, drought threatens (a) fish, (b) invertebrate and (c) macrophyte species of conservation concern (Defra 2022)—most likely alongside many diatoms and other microorganisms, for which rarity and conservation status have not been assessed (but see Kelly 2014a,b).

a. Drought threatens fish of conservation concern

From socioeconomic and legislative as well as ecological perspectives, fish are among England's most valued species to be threatened by drought. Some species, such as the sturgeon Acipenser sturio, are too rare in English rivers to enable investigation of their responses to drought. Drought puts populations of other species of conservation concern at risk:

- **Species at their thermal tolerance limits: salmon and grayling.** Atlantic salmon Salmo salar and grayling Thymallus thymallus are already at their thermal tolerance limits, leaving them particularly vulnerable to the elevated temperatures associated with concurrent droughts and heatwaves (Marsh et al. 2021; Rangeley-Wilson 2021). Other coldwater species including eel Anguilla anguilla, brook lamprey Lampetra planeri and river lamprey L. fluviatilis are also at risk.
- **Migratory species.** Diadromous species (which migrate between rivers and the sea, i.e. salmon, sea trout Salmo trutta morpha trutta, sea lamprey Petromyzon marinus and eel may be particularly susceptible to drought due to their

requirements for different, specific hydrological habitats at different life stages (Jonsson et al. 2007; Solomon and Sambrook 2004; Jonsson and Jonsson 2009; Robinson et al. 2009; Marsh et al. 2021). Such effects encompass the loss of the peak flows that trigger both upstream and downstream migration (Mann 1989; Solomon and Sambrook 2004); water depths that are too low to enable migration (Solomon and Sambrook 2004), to provide spawning habitats (Taverny et al. 2012; Pinder et al. 2016) and to provide refuge from predation (Riley et al. 2009); slow flow velocities that increase territory sizes (Symons and Héland 1978); and siltation that degrades spawning gravels (Berrie 1992; Riley et al. 2006). Consequences include reduced fecundity, recruitment, growth and survival as well as behavioural changes (Weatherley et al. 1991; Nislow et al. 2004; Mitchell and Cunjak 2007).

- The **twaite shad** Alosa fallax breeds in the Rivers Severn and Wye (JNCC 2022a) and requires deep pools in which to congregate prior to spawning, and fast-flowing, silt-free gravel-dominated spawning habitats (Aprahamian et al. 2003). Drought impacts on populations in English rivers have not been reported.
- The **spined loach** Cobitis taenia favours organic-rich sediments over gravel (Culling et al. 2003), prefers low-to-moderate flow velocities and higher densities of filamentous algae (Copp and Vilizzi 2004). These habitat preferences may enhance its drought tolerance.
- The **bullhead** Cottus gobio is at risk from elevated temperatures during drought (Dorts et al. 2012). But where temperatures remain moderate, bullhead can benefit from reduced competition for resources from juvenile salmonids during drought, increasing their survival (Elliott 2006).

The risk to fish in context. At the river network scale, dry phases fragment wet habitats, which can prevent fish movements including seasonal migrations to and from spawning grounds as well as drought-driven movements from shallow to deeper waters (Armstrong et al. 1998). As such, drought-driven network fragmentation by extreme low flows and dry reaches can reduce reproductive activity, recruitment and fish biodiversity (Perkin et al. 2015, 2019). The frequent, widespread interruption of upstream–downstream connectivity by artificial barriers such as dams in English rivers (Jones et al. 2019) exacerbates drought-driven network fragmentation and further reduces refuge-seeking fish movements.

If climate change increases the occurrence of drought:

- If flow declines earlier in the year and/or declines more quickly, and/or drought duration, magnitude and severity increase—any and all of these changes threaten fish populations, with migratory species being most at risk.
- Increases in drought severity that reduce water depths for longer periods will reduce availability of well-oxygenated, cool-water habitats for thermally sensitive fish, increasing summer fish kills.
- The capacity of drought-impacted fish populations to recover after a drought is compromised by the artificial barriers that fragment England's river networks.
- **b. Drought threatens invertebrates of conservation concern**
- Numerous British freshwater invertebrates are listed as threatened (JNCC 2022b), but we know too little about the ecological requirements of most such

species to confirm or predict their responses to drought. Just a few relatively high-profile species are highlighted below:

- **Crustaceans**

    - **White-clawed crayfish.** Providing a striking example of a permanent drought-driven population loss, the internationally Endangered white-clawed crayfish Austropotamobius pallipes was eliminated by drying of near-perennial reaches of Hertfordshire chalk streams (Ellis and England 2009; Perrow et al. 2007). Such dry phases may be particularly damaging to crayfish populations in chalk catchments due to their low stream densities, meaning that overland distances to wet refuges may be farther than crayfish can walk (Masefield 2018). This loss illustrates that drought-driven river drying can interact with other pressures (here including groundwater abstraction and the non-native signal crayfish Pacifastacus leniusculus) to cause local and regional extinction of a species of conservation concern (Perrow et al. 2007).

    - **Groundwater specialists.** Several British ‘stygobitic’ crustaceans are of conservation concern, but we know little about their responses to drought. Declining groundwater oxygen concentrations during concurrent drought and heatwave events can trigger migrations of stygobitic Niphargus shrimps upwards into the hyporheic zone (Wood et al. 2010; Stubbington and Wood 2013). Looking deeper, Durkota et al. (2019) studied stygobites in the chalk aquifer during a drought in southern England (Met Office 2013), detecting responses for species including Crangonyx subterraneus, which was more abundant during the drought peak, suggesting refuge-seeking behaviour (J. Durkota, pers. comm.).

- **Insects**

    - **The southern damselfly.** Drying of typically inundated marginal sediments can kill desiccation-sensitive immotile life stages including eggs, of species including the internationally Endangered southern damselfly Coenagrion mercuriale (Daguet et al. 2008). This rare species requires perennial flow and dry phases could eliminate it from key habitat types including small heathland streams (Purse 2002).

    - **Winterbourne specialists.** The Nationally Rare winterbourne stonefly Nemoura lacustris and the Nationally Scarce scarce purple dun mayfly Paraleptophlebia werneri are largely restricted to chalk winterbournes (Macadam 2015, 2016), which may be due to the lower densities of their competitors (in particular, Gammarus shrimps; Aspin and House 2022). Gammarus recolonize winterbournes by swimming upstream after flow returns (Wood and Petts 1994) and winterbourne specialists thus thrive in reaches far enough upstream to exceed the distance that Gammarus can move during a wet phase (Aspin and House 2022). As such, drought-driven increases in dry-phase durations could increase the spatial length of winterbourne in which specialists can occur—unless drought also causes downstream migration of winterbourne heads.

  - **Other insects.** Identifying species of conservation concern within drought-sensitive families could reveal a much greater range of species threatened by drought. For example, eight species within the most drought-intolerant families identified by Chadd et al. (2017) are of conservation concern (Appendix 3). Research is needed to identify priority species for which action is required to protect populations at risk of extinction.

– **The pearl mussel.** The internationally Endangered freshwater pearl mussel Margaritifera margaritifera is long-lived and relatively immotile, increasing the risk posed by drought. Mussels can be stranded in drying marginal sediments as flow declines, in particular when rapid water loss prevents small-scale movements into wet habitats (Curley et al. 2022). Stranded mussels may die due to either desiccation or predation (Sousa et al. 2018), and resultant population declines could reduce recruitment of future generations.

If climate change increases the occurrence of drought:

- Any increase in drought will—in interaction with other human pressures—exacerbate regional extinction risks for invertebrate species including the white-clawed crayfish and freshwater pearl mussel.
- We know little about how most invertebrate species—including insects and groundwater crustaceans of conservation concern—may respond to future drought.
- Although it is feasible that drought impacts on common species could create space in which rare species thrive, it is more likely that the cumulative impacts of longer, more severe, more frequent drought could drive rare invertebrates closer to extinction.

c. Drought threatens macrophytes of conservation concern.

Ranunculus. Several species in the water crowfoot Ranunculus subgenus Batrachium characterize river types protected as Special Areas of Conservation under the Habitats Directive (JNCC 2022c). The negative effects of drought on Ranunculus can be considerable, as discussed in section 6.

If climate change increases the occurrence of drought:

- Any increase in drought is likely to reduce growth and survival of the water crowfoot Ranunculus subgenus Batrachium.

12. Droughts: effects on non-native invasive species

Drought-driven **low flows** may enable current UK invasive non-native species (INNS) to become more prevalent. For example:

- Low flows can promote the establishment of **signal crayfish** Pacifastacus leniusculus at newly invaded sites, leading to range expansion during years with below-average flows (Mathers et al. 2020).
- The **New Zealand mud snail** Potamopyrgus antipodarum prefers stable flows (Bennett et al. 2015) and can thrive during low flows due to high availability of the biofilms upon which it grazes (Wood and Petts 1999). Its population densities can thus increase during low-flow years (Wright 1992), with high-density populations then persisting after normal flow resumes (Wood and Petts 1994, 1999).
- Low flows may facilitate the ongoing range expansion of the **demon shrimp** Dikerogammarus haemobaphes through central England (Johns et al. 2018).

Dry phases. Drought-driven, partial or complete surface water loss may favour INNS that better tolerate desiccation compared to equivalent native species. For example:

- **Signal crayfish** can survive in air at 24°C for <21 hours (Banha and Anastácio 2014), with this tolerance exceeding that of the native white-clawed crayfish Austropotamobius pallipes (Holdich et al. 2014). Signal crayfish can nonetheless be eliminated by rare dry phases in near-perennial reaches, but may soon return, whereas natives may be permanently lost (Perrow et al. 2007).
- The **New Zealand mud snail** tolerates some desiccation, with closure of its operculum facilitating survival during short (days to weeks) dry phases (Storey and Quinn 2011; Lancaster and Ledger 2015). However, its population densities may decline substantially if dry periods persist for weeks to months (Extence 1981).
- Drought, in particular prolonged drought during summer (i.e. the growing season), may alter the distribution of riparian plants including the INNS **Japanese knotweed** Fallopia japonica and **Himalayan balsam** Impatiens glandulifera. Future northward range contraction has been suggested for both knotweed (Beerling et al. 1995; Hulme 2017) and balsam (Beerling 1993), reflecting water requirements not being met due to climatic drying. However, whilst rising temperatures will drive increases in evapotranspiration that reduce soil moisture, especially in the south, changes in precipitation remain uncertain (Samaniego et al. 2018; Shaffrey 2023). As such, any near-future range contractions may be localized and most likely to occur during summer droughts (Beerling 1993).
- **Rainbow trout**. Loss of fish due to drought-driven dry phases can be compensated by both legal stocking and illegal releases after flow returns. The latter may reintroduce INNS to sites from which they have been eliminated by dry phases and introduce them to new sites, as occurred for rainbow trout Oncorhynchus mykiss in the River Misbourne (Perrow et al. 2007).

13. Drought can simplify food webs and ecosystem processes

Drought-driven changes in the occurrence, abundance, densities or biomass of individual species can alter food web structure and function, including 'bottom-up' effects that result from changes to the resources at the base of both green (i.e. plant and biofilm-based) and brown (i.e. detritus-based) food webs and 'top-down' effects that start with changes to predator populations at higher trophic levels.

Bottom-up effects on green food webs. Drought-driven low flows can increase light and nutrient availability, causing short-term increases in primary production by biofilms at the base of food webs. As a result, densities of (and secondary production by) primary consumers, i.e. grazer invertebrates could increase (see section 5), with knock-on effects at higher trophic levels. Equally, low flows can reduce river-scale primary production if wetted habitats contract, killing or reducing the metabolic activity of microbial biofilms exposed to air (Truchy et al. 2020; Arias Font et al. 2021). If drought causes streams to dry completely, drastic reductions in biofilms reduce primary production and thus the biomass available to sustain organisms (including early colonists) at higher trophic levels after flow resumes, which could limit initial rates of ecosystem recovery (Ledger and Hildrew 2001).

Bottom-up effects on brown food webs. The dominant invertebrate in many English rivers is a 'shredder', the shrimp Gammarus pulex/fossarum (Macneil et al. 1999). Its shredding activity breaks down and transforms detritus such as leaf litter, creating diverse energy sources that can be used by organisms at lower trophic levels (Gessner et al. 1999). In addition, Gammarus is an important prey species for predators at higher trophic levels including other invertebrates, fish (Macneil et al. 1999) and riverine birds (Shaw 1979). Gammarus are desiccation sensitive and are much reduced or eliminated by dry phases, reducing detrital decomposition as well as trophic resources for predators, thus reducing secondary production and energy transfer through brown river food webs and to terrestrial ecosystems (Datry et al. 2011; Ledger et al. 2011; Vander Vorste et al. 2016b).

Top-down effects of predator loss. Drought-driven dry phases can greatly reduce the densities and dominance of large-bodied invertebrates, in particular predators such as juvenile dragonflies (Nelson et al. 2021) and alderflies (Ledger et al. 2011). These losses alter food web structure during subsequent flowing phases (Woodward et al. 2012; Ledger et al. 2013), reducing trophic interactions, total biomass and secondary production, and simplifying food webs as species are lost (Ledger et al. 2011, 2013). Release from predator control could increase the abundance of small-bodied invertebrates (such as Chironomidae midges) at intermediate trophic levels, partially compensating for the reduction in secondary production, but this may be insufficient to maintain pre-drought rates of energy transfer (Ledger et al. 2013).

Drought may also directly eliminate taxa at intermediate trophic levels. For example, filter feeders including Simuliidae blackfly larvae are important both as primary consumers and as prey (Ladle et al. 1972; Mann and Blackburn 1991), but die if their food supply is cut off by flow cessation (Sarremejane et al. 2021a).

14. Recovery after drought

Few studies reporting ecological recovery after drought have sufficient pre-drought data to evaluate return to a pre-drought state (Table 2). In addition, characterization of recovery is limited by study duration, with relatively few studies reporting patterns over multiyear timescales (Table 2)—and those that do are typified by a coarse temporal resolution i.e. collection of 1–2 biological samples per year (e.g. Wright [1992] and others in Table 2). In any case, single, stable baseline states can be difficult to characterize in naturally dynamic river ecosystems (Thoms 2006; Ryo et al. 2019). Climate change is compounding these difficulties, and increasingly undermines the notion that ‘natural’ ecosystems fluctuate within unchanging boundaries (Milly et al. 2008). Despite these caveats, post-drought community recovery can be, and has been, characterized—largely for macroinvertebrates (Table 2), a unique group in that its recovery may be supported by overland flight of adult insects (Dobel et al. 2020).

Table 2. Summary of UK studies examining recovery of ecological communities after drought. Studies are presented in chronological order, by year of publication. Drought periods are as stated by the authors. EPT, Ephemeroptera, Plecoptera and Trichoptera; MIV, aquatic macroinvertebrates; ↓, decrease; ↑, increase. 'Permanent' is limited by the study duration.

Study system and site(s)	Drought period	Study start and end date	Post-drought period	Habitat conditions during drought	Biotic group	Peak biotic impact of drought	Evidence of recovery from drought	'Permanent' taxon losses; new taxon gains	Source
Afon Hirnant, a small mountain stream, Wales. **3 sites studied but the study focuses on 1 site that dried.**	Very dry summer 1955. Flow normal Nov 1955.	"Regular" sampling from Oct 1954 to an unspecified date in 1956	<1 year i.e. Nov 1955 to 1956	Rare, complete dry phase affecting 1 km from early August to mid-Oct 1955	Fish MIV Plants	Total loss of brown trout Total loss from surface channel Survived in most places	Recolonized from down-stream in Jan. Rapid recovery from subsurface sediments Grew rapidly when water returned	9 EPT spp. did not return; *Nemoura cinerea* appeared	Hynes 1958
Winterbourne tributary of the chalk River Lambourn, Berkshire. **Possibly 4 sites; unclear.**	1976 summer drought	"Throughout 1972, during the drought of 1976 and … to August 1977"	<1 year i.e. to August 1977	1 site flowing, 1 site dry with "ponds connected by ditches	MIV	Taxa richness ↓ "dramatically"	Most species back to their 1972 distribution but not abundance by summer 197[7]	Many opportunist species had not recolonized by the end of the study	Freshwater Biological Association 1977–79

Waterston stream, a small, near-perennial winterbourne chalk stream. **A 50-m stretch.**	"A long period with low rainfall" in 1973; flow resumed in Jan 1974	MIV: Approx. monthly Mar–Oct 1973 + Jan–Aug 1974, i.e. no pre-drought data. Plants: Monthly Jan 1973–July 1974	<1 year, i.e. 1974	Rare dry phase	MIV Plants	Composition changed: *Gammarus* ↓ and *Serratella* ↑ Composition changed: *Ranunculus penicillatus* var. *calcareus* ↓ and *Apium* ↑	Fecund spp. e.g. *Asellus* recovered in <1 year	*Polycelis felina* and *Agapetus fuscipes* not recorded 1974; *Gammarus* almost eliminated. *Ranunculus peltatus* colonized upstream of site.	Ladle and Bass 1981
Afon Dulas, an upland tributary of the Severn, Wales. **1 site.**	1976 summer drought	MIV: Monthly from March 1976 to Sept 1977 + follow-up surveys in 1978–79. Fish: 10 dates from Oct 1975 to April 1978.	MIV: 3 years, i.e. 1977–79 Fish: 2 years, i.e. 1977–78.	Low flow (width, depth) and exceptionally high water temperatures	Fish MIV	Loss of 1976 salmon year class. Abundance 40% of 'normal', many insect larvae died, composition changed.	Higher salmon recruitment in 1977. MIV community diversity and abundance fully recovered in 1978–79.		Cowx et al. 1984

4 small forested, acidic streams in central Scotland. **1 site per stream.**	Summer 1984, including low rainfall in May to August.	Approx. monthly sampling from Sept 1984 to March 1985 i.e. no pre-drought data	6 months i.e. to March 1985	3 streams “dried out almost completely” for 2–3 months; 1 “reduced to a series of small pools”. Subsurface water present.	Fish	Dead trout on dry streambeds. Some trout survived in pools.	Spawning trout entered streams after drought ended.		Morrison 1990
					MIV	Dead insects on dry streambed.	Small larvae from several orders <1 month after flow returned. Abundance of most taxa similar pre- and post-drought.	Several taxa recorded only pre-drought, a few recorded only post drought; no notable losses	
The chalk River Lambourn and its tributary, the Winterbourne stream, Berkshire. **30 sites in total; 2 perennial sites for 1971–78 study**	Minor drought 1973, severe drought 1976	June and December surveys in 1971–78	2 years: to 1975 for 1973 drought, and to 1978 for 1976 drought	Low flows at perennial sites. Unusually high levels of deposited silt.	MIV	Very high chironomid densities during low flows	No substantial community change pre-and post-drought	None reported	Wright 1992; also see Wright and Symes 1999
					Plants	Poor growth of *Ranunculus*	1–2-year recovery periods		
				Very long (16-month) dry phases at winterbourne sites	MIV	Low taxa richness	Rapid recolonization from perennial sites		
					Plants	*Ranunculus* survived			

15 sites along the chalk River Little Stour, Kent	Autumn 1988–late 1992	Annual sampling; autumn 1992 and 1993 i.e. no pre-drought data	<1 year i.e. autumn 1993	Sustained, severe low flows (width, depth); upper reaches dried. Silt and sand deposition.	MIV	Taxa richness slightly lower and abundance 5-fold lower in 1992: most taxa survived at low densities	Increased abundance in 1993 indicate rapid recovery after normal flow resumed	Insufficient data	Wood and Petts 1994 (Also see Wood and Petts 1999)
8 sites on chalk Rivers Test and Itchen	Hot dry summers: 1990,1994 Very low flows: 1991,1992	17 dates between July 1991 and Jan 1996	2 years i.e. 1993–94	Low flows including slow velocities and reduced depth	Plants	*Ranunculus* ↓ and filamentous algae ↑	*Ranunculus* looked “abundant and vigorous” in 1994	Not reported	Wilby et al. 1998
118 sites on 24 chalk stream reaches, south England	Poor groundwater recharge 1989–92	1992 and 1995 i.e. no pre-drought data	<4 years (i.e. 1992–95)	Low flows at perennial sites; long dry phases at winterbourne sites	Plants	Flow drove compositional changes. Taxa-rich communities in more natural channels more altered by drought.	Communities recovered in <2 years	Not reported	Holmes 1999; also see Westwood et al. 2006a
14 sites along the chalk River Little Stour, Kent	Autumn 1988–late 1992, plus summer rain deficit in 1995	Autumn 1992 and 1995 i.e. no pre-drought data	3 years (i.e. 1993–95)	Low flows at perennial sites; rare dry phases in near-perennial sites; long dry phases in	MIV	Low abundance and reduced richness in 1992 compared to later years	Richness ↑ from 58 taxa in 1992 to 79 in 1995; abundance ↑ from 64 individuals in	A few taxa eliminated, i.e. *Caenis robusta* not recorded after 1992–93	Wood and Petts 1999

				winterbourne sites			1992 to 1785 in 1995.		
15 sites along the chalk River Little Stour, Kent	Autumn 1988–late 1992, then 1996–97	1992–97 i.e. no pre-drought data	3 years after 1992 drought i.e. 1993–95	As for Wood and Petts (1999), above	MIV	As for Wood and Petts (1999), above	As for Wood and Petts (1999); suggestion that recovery takes 2 years (but no pre-drought data)	Not reported	Wood et al. 2000
Lone Oak, a first-order, acidic, near-perennial forested stream, SE England. **1 site.**	Summer 1995. High flows in winter 1995–96	Aug 1994 to March 1996 i.e. including a pre-drought period from Aug 1994 to March 1995	<1 year i.e. Sept 1995 to Feb 1996	Surface stream dried from July to Sept 1995	MIV Biofilm	Low densities at end of dry phase; composition temporarily altered by rapid colonization of pioneers Reduced chlorophyll and algal densities	High flows reset community to its pre-drought composition in winter 1995–96. Cells persisted during dry phase, enabling rapid recovery to pre-drought conditions	*Simulium* not recorded post-drought. *Stempellinella*, *Polypedilum* and *Paraleptophlebia* first recorded post-drought.	Ledger and Hildrew 2001
The chalk River Kennet, Wiltshire. **3 sites.**	1996–autumn 97	4 survey dates: July 1997, Dec 1997, June 1998, June 1999	18 months i.e. Dec 1997 to June 1999	Low flows and associated silt deposition	MIV Plants	Compositional change: lentic taxa favoured over lotic taxa. *Schoenoplectus* ↓, *Ranunculus* ↓ and *Callitriche* ↑	Composition rapidly returned to ‘normal’. *Ranunculus* recolonized rapidly.	None reported	Wright et al. 2002, 2003

3 sites on the chalk River Kennet, Wiltshire, and 1 on the chalk River Lambourn, Berkshire	1996–autumn 97	Summer 1997 to summer 2001: no pre-drought data, extends Wright et al. 2002 post-drought data	4 years i.e. summer 1998–2001	As above for Wright et al. 2002, 2003	MIV	As above for Wright et al. 2002, 2003	As above for Wright et al. 2002, 2003. Limited faunal change in 1998–2000 indicates that recovery was complete by 1998	None reported	Wright et al. 2004
15 sites along the chalk River Little Stour, Kent	1991–92, 1996–97 and 2005–06	Annual samples in Aug/Sept 1992–99 then monthly April–Oct 2006	3 year post-1992, 2 year post-1997, none in 2006	As for Wood and Petts (1999) in 1991–92 and 1996–97. Severe low flows 2005–06	MIV	Abundance ↓ and richness ↓ especially insects, coincident with peak temperatures. Early insect emergence.	Abundance ↑ and richness ↑ over 2 years Highest taxa richness in 1995, 3 years post-drought	None reported	Stubbington et al. 2009a
46 sites on 7 chalk streams, Hertfordshire. **Focus on near-perennial and winterbourne sites**	Based on no-flow days: winter 1996–97, summer 2005–06, winter 2012, summer 2017	Samples collected in spring and autumn 1995–2017	8 years post 1996–97, 5 years post 2012	Rare dry phases in near-perennial sites; long dry phases in winterbourne sites	MIV	LIFE (Extence et al. 1999) and DEHLI (Chadd et al. 2017) scores ↓ as summer no-flow days ↑ especially at near-perennial sites	DEHLI (Chadd et al. 2017) scores took 3 years to recover after drying of near-perennial sites. Taxon-specific increases in occurrence took 60 d to 10 years	Nemouridae and Sericostomatidae required >10 years of continuous flow to recover	Sarremejane et al. 2019
30 perennial, near-perennial and	2005–07 and 2011–12	Biotic data collected in spring 2006–	3 years post-2007 and 5	Low flows at perennial sites; dry phases in	MIV	Local taxa richness ↓ with extent of drying	Local taxa richness ↑ and spatial β diversity	None reported	Sarremejane et al. 2020

winterbourne sites across 7 chalk streams, Hertfordshire		17; hydrological data from 2004–18	years post-2012	near -perennial sites; long dry phases in winterbourne sites		No change in regional richness	↓ as flowing phase duration increased, indicating rapid recovery		
20 perennial, near-perennial and winterbourne sites across 7 chalk streams, Hertfordshire	2005–06 and 2011–12	Biotic data from spring and autumn 2005–17; hydrological data from 2004–18	4 years post-2006 and 5 years post-2012	As for Sarremejane et al. 2020	MIV	Risk of population loss ↑ as regional extent of flow ↓ Risk of population loss ↑ after 3 drought years	Alternating periods of drought and normal flows kept risk of population loss low	Not reported	Sarremejane et al. 2021
1 site on the South Winterborne, a small winterbourne chalk stream, Dorset	Autumn–winter 2011–12, then high flows led to 13 months continuous flow	Annual sampling in 2009 to 2020	Several years	Unusually long (estimated 10-month) dry phase; flow did not resume until May 2012	MIV	Not characterized: samples not collected due to drying	Community resembled that at a perennial site in 2013 then reverted to a winterbourne community in 2014	None reported	Bass et al. 2022 and sources therein

Communities—but not all species—recover from drought within 1–3 years. Across English rivers, communities are typically resilient to even severe drought-driven conditions (e.g. a rare dry phase in a near-perennial reach), with biofilms, macrophytes, invertebrates and fish often recovering within 2 years at both perennial and temporary sites. For example (these and other examples are summarized in Table 2):

- **Algal** and **invertebrate** communities recovered within weeks of flow resuming after a 9-week, near-complete, rare, drought-driven dry phase in a low-nutrient, acidic, near-perennial, forested stream in south England (Ledger and Hildrew 2001). Post-drought taxonomic changes only persisted until high winter flows restored pre-drought community composition. Similar patterns have been reported in acidic, forested small streams in central Scotland (Morrison 1990) and a small upland stream in Wales (Hynes 1958). At these sites, rapid recovery may reflect the stream's natural communities, which comprise a relatively small number of stress-tolerant, disturbance-adapted species.
- A **brown trout** Salmo trutta population (and an invertebrate community) recovered within 2 years of a drought-driven low-flow period in a small upland stream, despite loss of an entire trout age class and despite a lack of upstream sources of colonists (Cowx et al. 1984). Recovery of the trout population may have been enabled by reduced territorial aggression and thus increased juvenile survival after the drought, whilst recovery of the insect-dominated invertebrate community likely reflected recolonization following egg-laying by adult insects.
- Holmes (1999) reports the recovery of **macrophyte** communities in headwater chalk streams affected by a wide range of drought-driven hydrological changes, from low flows to long dry phases. Despite site-specific and species-specific variability, most communities recovered in 1–2 years.

After a rare dry phase, communities at drought-impacted near-perennial sites can take at least 3 years to return to a composition comparable to that of communities at perennial sites (Wood and Petts 1999; Sarremejane et al. 2019). Taxa with 'resilience' traits such as short lifecycles, including Baetidae mayflies and Simuliidae blackflies, can recolonize rapidly (Wright and Symes 1999; Ledger et al. 2011; Sarremejane et al. 2019, 2020). However, such invertebrates use water crowfoot (Ranunculus species) as habitat, and their population recovery can thus be delayed until Ranunculus returns to pre-drought levels, which can take >1 year (Wright 1992). In addition, taxa with longer (e.g. 2-year) life cycles, such as the green drake mayfly Ephemera danica and the caddisfly Sericostoma personatum can take <10 years to recover (Wright and Symes 1999; Ledger et al. 2011; Sarremejane et al. 2019).

Communities recover by recolonization from local to network-scale refuges. During drought, all English river networks retain extensive perennial reaches at the catchment scale; at the river scale, each system has sufficient morphological variability to retain some fast-flowing, deeper refuge habitats; and at the site scale, wet and damp refuges may persist in even the most drought-impacted reaches (see Fig. 6, section 16). This environmental variability can promote asynchronous fluctuations in local-scale species-specific population densities during drought, limiting concurrent population declines and promoting long-term network-scale metapopulation persistence (Sarremejane et al. 2021b).

After a drought ends, populations persisting in refuges can supply colonists to impacted sites. Connected, flowing reaches can supply fully aquatic species such as Gammarus shrimps, and as a result, river-scale recovery rates relate positively to the spatial extent of flowing reaches that persist during drought (Sarremejane et al. 2020). Riparian and terrestrial vegetation also provides habitat for adult insects that recolonize by flight (Bogan et al. 2017). In addition, within-site refuges such as pools and saturated subsurface sediments can support recovery of even sites with poor connectivity to other, off-site refuges (Hynes 1958; Vander Vorste et al. 2016a).

Recovery depends on site location. The rate and extent of local community recovery depend in part on the distance and direction to drought-impacted sites from colonist sources as well as connectivity between these sites and sources (Driver and Hoeinghaus 2016; see Fig. 6, section 16). As such, site-specific recovery rates depend on a site's position in a river network. Where drought-driven dry phases occur in mid-reaches downstream of perennial surface waters and/or saturated subsurface sediments, drift may supply copious colonists that contribute to community recovery as soon as flow resumes (Wood and Petts 1994, 1999; Pařil et al. 2019; Fournier et al. 2022). With progression upstream, isolation from colonist sources typically increases and temporary headwater streams typically lack upstream perennial reaches, eliminating drift as a source of post-drought colonists. Communities in headwater sites may thus take longer to recover from drought (Tornwall et al. 2017). However, in groundwater-fed headwater chalk streams, water can slowly return from downstream to upstream as the water table rises, allowing immediate recolonization of newly wet reaches by organisms within the wetted front (Moon 1956).

Communities recover faster in more natural streams. Ecological recovery from drought takes longer in rivers subjected to other human pressures. In particular, the extensive occurrence of dams and other artificial barriers throughout England's river networks (Jones et al. 2019) reduces ecological resilience, because it prevents motile organisms including fish and invertebrates that persist in catchment-wide refuge sites from accessing impacted sites after a drought ends (Perrow et al. 2007). In addition, physical modification homogenizes in-channel habitats, reducing the range and extent of local refuges in which organisms can persist throughout a drought (Dunbar et al. 2010a,b; see Fig. 6, section 16).

For example, post-drought recovery trajectories of macrophyte communities in headwater chalk streams include slow, partial recovery of species-poor communities in over-abstracted, modified channels but complete, rapid recovery of species-rich communities in more natural sites (Westwood et al. 2006a,b). Similarly, Wright et al. (2003) attributed rapid post-drought recovery of invertebrate communities in a perennial section of the River Lambourn to its natural habitat characteristics; the same likely applies after dry phases in previously perennial, near-perennial and other temporary streams.

If climate change increases the occurrence of drought:

- Any increase in drought frequency could prevent recolonization by less resilient taxa, risking local species extinctions and reducing community biodiversity.
- Any increase in drought severity could reduce the types and extent of refuges, limiting metacommunity persistence and thus post-drought community recovery.
- Any increase in the spatial extent of drought will increase recovery times for communities in isolated headwaters, with concurrent increases in drought frequency potentially preventing complete recovery.
- Any increase in drought is likely to have particularly severe impacts on communities at physically modified sites and those isolated from refuges by artificial barriers.

15. Tipping points: Drought pushes ecosystems to new, altered states

Studies reporting that communities typically recover from drought within weeks to a few years date back as far as notable events in 1955 and 1976 (Table 2). We cannot assume that ecological responses to these events and recent droughts will represent responses to current and future droughts, given the expected increase in drought frequency, severity and spatial extent (Hall et al. 2022; Lane and Kay 2023)—and thus drought extremity (Smith 2011). In addition, future droughts (including their climatic and artificial drivers) will inevitably interact with rising temperatures and multiple human pressures to determine ecological responses (Van Loon et al. 2016). In short, future droughts could have greater, longer-lasting ecological impacts that those documented to date (Crausbay et al. 2020).

Drought may cause permanent shifts to new ecosystem states. There is an increasing risk that future droughts will be truly extreme events (sensu Smith 2011) and thus be ecologically ‘transformative’ (sensu Crausbay et al. 2020), pushing ecosystems past tipping points i.e. thresholds at which their structure and function shift to a new ‘stable’ state (Scheffer et al. 2001; van Nes et al. 2016). As naturally dynamic, disturbance-prone ecosystems, rivers are less vulnerable to state shifts than other ecosystems. Nonetheless, shifts do occur in rivers, for example from dominance of phytoplankton to macrophytes due to reduced phosphorus concentrations and increased phytoplankton consumption by an invasive clam (Minaudo et al. 2021; Diamond et al. 2022); and from dominance of filamentous algae to macrophytes following livestock removal (Heffernan 2008; Ribot et al. 2022). Drought-driven state shifts could thus occur in river ecosystems, and would be likely to reduce biodiversity and simplify ecosystem structure and function (Ledger et al. 2013).

Flow regimes are changing, and changes may be permanent. Shifts in flow regimes—for example to near-perennial flow at previously perennial sites, to higher dry-phase frequencies at near-perennial sites, and to longer dry-phase durations at seasonally intermittent sites—are occurring, are expected to continue, and are likely to have ecosystem-scale effects (Tramblay et al. 2021; Zipper et al. 2022). Such hydrological shifts seem like likely triggers of drought-driven state shifts, but shifts may be localized and incremental. For example, in space, a shift from perennial to near-perennial flow may occur over a short stream length, or in time, average drying frequencies could increase from e.g. every 10 to every 8 years. Nonetheless, these localized, incremental shifts collectively represent a large-scale reduction in aquatic habitat availability during drought (Sarremejane et al. 2020).

For now, the modest spatial extent of drought-driven drying may enable post-drought recovery in England's river networks (Sarremejane et al. 2021b). But ecological resilience is likely to decline if drought conditions—and in particular, dry phases—increase in space and/or time. As such, multiyear droughts have caused persistent changes in hydrological regimes and thus ecosystem structure and function in Australia and the US (Peterson et al. 2021; Fowler et al. 2022). Similarly, a network-scale increase in the spatial extent of drought-driven drying could increase local-to-regional-scale extinction risks of desiccation-sensitive species (Jaeger et al. 2014), with permanent consequences for ecosystem structure and function depending on the species lost, as discussed below.

Extinctions have limited impact unless key species are lost. Stochastic and deterministic local-scale species losses inevitably occur during drought (see section 7), and can include permanent local extinctions (Perrow et al. 2007). The loss of individual species reduces biodiversity but typically has limited impact on ecosystem functioning due to 'functional redundancy' (i.e. multiple species having the same traits, such as the same mode of feeding). However, when a key species is lost—be that an abundant, dominant species, a habitat-forming species (McIntosh 2019) or a keystone species (i.e. one which has a disproportionate effect on ecosystem structure and function given its abundance; e.g. Power et al. 1996)—ecosystems may shift to new states. In addition, each single drought-driven species loss contributes to a slow, incremental decline in functional redundancy, increasing the risk that the next species loss will eliminate a particular trait and thus reduce ecosystem functioning (Fonseca and Ganade 2001).

– **Key species include beavers, salmon, bullhead, shrimps and water crowfoot.** In English rivers, key species include beaver Castor fiber, active reintroduction and management of which mitigates any risk to its persistence posed by drought (Brazier et al. 2021); Atlantic salmon Salmo salar, brown trout Salmo trutta and other large-bodied predatory fish, which may be rescued and/or stocked to support their persistence despite drought (Woodward et al. 2021); the small-bodied fish bullhead Cottus gobio, an abundant predator and prey species (Woodward et al. 2008) which is legally protected and thus also rescued during drought; the shrimp Gammarus pulex/fossarum, an abundant shredder of leaf litter (Woodward et al. 2008); and the macrophyte water crowfoot (Ranunculus species), which provides extensive habitat for species including epiphytic algal, invertebrates and fish, and which alters flow and sediment dynamics (Gurnell et al. 2006). Many other abundant, dominant species may also influence ecosystem functioning. For example, Chironomidae midges and snails can be highly abundant grazers of algal biofilms, creating a 'keystone interaction' which may influence an entire food web (Schaum et al. 2017).

- **How might an ecosystem change if a key species is eliminated by drought?** Its loss will create habitat space which another species can exploit, or the niche space may be left empty. The consequences of each loss are species-specific, site-specific and hard to predict, but the following scenarios could trigger a state shift:

  - **Local extinction of a top predator** such as a fish or large-bodied invertebrate is a likely consequence of drought (Ledger et al. 2013). This loss (unless reversed by fish stocking) could cause top-down 'trophic cascades' (i.e. effects at multiple trophic levels within a food web) by releasing prey from predator control, allowing their abundance to increase. For example, Gammarus shrimps and Potamopyrgus mud snails can be more abundant when bullhead are absent, potentially increasing both leaf litter decomposition (by shredder shrimps) and biofilm consumption (by grazing snails; Woodward et al. 2008). Overall food web structure and function would be left permanently simplified (Ledger et al. 2013).

- **Loss of Ranunculus**. Drought can cause the loss of habitat-forming macrophytes such as Ranunculus due to slow flow velocities and associated silt deposition (see section 6). The direct consequences of its loss are to create niche space, which is likely to be filled by slow-flow-loving filamentous algae such as Cladophora glomerata and emergent macrophytes such as fool's watercress Apium nodiflorum. A bottom-up trophic cascade ensues, including the loss of invertebrates such as Baetidae mayflies and Simuliidae blackflies, then an associated reduction in their predators, including fish. In addition, the change from dominance of Ranunculus, a submerged macrophyte, to filamentous algae will alter flow patterns, and thus sediment transport and deposition, and thus river planform (Cotton et al. 2006; Dobel et al. 2020).

- Although the return of suitable flow conditions can enable Ranunculus to re-establish populations, an increase in drought frequency and severity could permanently shift producer communities from macrophyte to algal dominance. Any such shifts will reflect a balance of multiple natural and anthropogenic stressors: the predicted increase in summer storms (Arnell et al. 2015) and higher winter flows (Watts et al. 2015) could favour Ranunculus over Cladophora, and inorganic nutrient concentrations, sediment composition, water depth, shading and invertebrate grazing will also influence the outcomes of competitive interactions between the two genera (Wilby et al. 1998).
- **Loss of Gammarus shredders**. The common, often abundant shrimp Gammarus pulex/fossarum may decline during drought-driven low flows and may be temporarily eliminated by dry phases (Stubbington et al. 2009b; Datry et al. 2011). As a result, leaf litter processing and associated secondary production may decline, even if other shredders such as the hoglouse Asellus aquaticus are present (Iversen et al. 1978; Chergui and Pattee 1988), as will prey availability for predatory invertebrates and fish. The consequences of its absence could extend throughout food webs.
- However, Gammarus is motile, has a tendency to swim upstream, can swim through surface water and burrow through subsurface sediments, and thus quickly recolonizes after water returns (Hill et al. 2019). As such, any shift in ecosystem state driven by its loss is likely to be localized, for example occurring in headwater reaches that are far enough from downstream colonist sources to prevent recolonization before the next dry phase.

Riparian vegetation loss could alter food web structure and ecosystem functioning. Riparian vegetation is sensitive to both soil moisture and groundwater levels (Garssen et al. 2014) and drought—in particular events that occur during summer growing seasons—can kill a high proportion of riparian trees, including up to 100% of saplings (Webb and Leake 2006; Bond et al. 2008; Giling et al. 2009; Reich et al. 2023; UK examples are lacking, but Kirby et al. 1998 report striking non-riparian losses). Reduced riparian canopies increase light availability and temperatures, supporting higher primary production by biofilms and plants whilst reducing leaf litter as an energy source (Giling et al. 2009). Anthropogenic factors such as inorganic nutrient concentrations as well as drought-related conditions could influence whether algae or vascular macrophytes dominate. The consequences of altered basal resources will have bottom-up effects that extend through food webs. Any permanent shift to an alternative stable state will depend on the recovery of riparian vegetation.

If climate change increases the occurrence of drought:

- Dry years tend to cluster together (Cole and Marsh 2006), resulting in multiyear droughts, such as occurred in England in 1975–76, 1989–92, 2004–06, 2010–12, and much longer historic events include the 1890–1910 “Long Drought period” (Marsh et al. 2007; Barker et al. 2019). This historic context, the spectre of multiyear events such as Australia’s 2001–09 Millennium Drought and the emerging North American megadrought (sensu Woodhouse and Overpeck 1998; Williams et al. 2020), intensifying climate change, and the emerging concept of drought as a water deficit that results from both climatic and human influences (Van Loon et al. 2016) collectively increase the likelihood of unprecedented drought events. Long multiyear events are particularly likely to cause ecosystems to shift to alternative stable states (Peterson et al. 2021; Fowler et al. 2022).

- Any increase in drought frequency could cause species-specific recolonization times to fall short of diminishing between-drought intervals, and where key species fail to return, ecosystem state could change. Loss of habitat-forming beds of Ranunculus has particular potential to cause shifts from macrophyte to algae-dominated communities.
- Increases in drought frequency and severity will cause additional local extinctions which collectively reduce functional redundancy (Leigh et al. 2019), pushing ecosystems closer to tipping points at which ecosystem functioning is altered.
- Predicting ecosystem-scale changes is tricky, and climate change is taking us into uncharted territory, increasing the risk of drought-driven ‘ecological surprises’ (see Filbee-Dexter et al. 2017).

16. Managing rivers to promote drought resilience

Extensive river regulation, intensifying climate change and increasing water resource pressures all alter the flow regimes of England’s rivers. In this context, updating the Environment Agency’s (2017) description of drought as “part of the natural water cycle”, we arguably can no longer accept drought as a natural phenomenon which ecosystems can cope with and recover from (Van Loon et al. 2016). Instead, river managers must act to maintain and enhance ecological resilience to drought in a context of multiple pressures (Scheffer et al. 2001; Tonkin et al. 2019). Figure 6 highlights a range of factors that managers can manipulate to promote ecological resilience to drought.

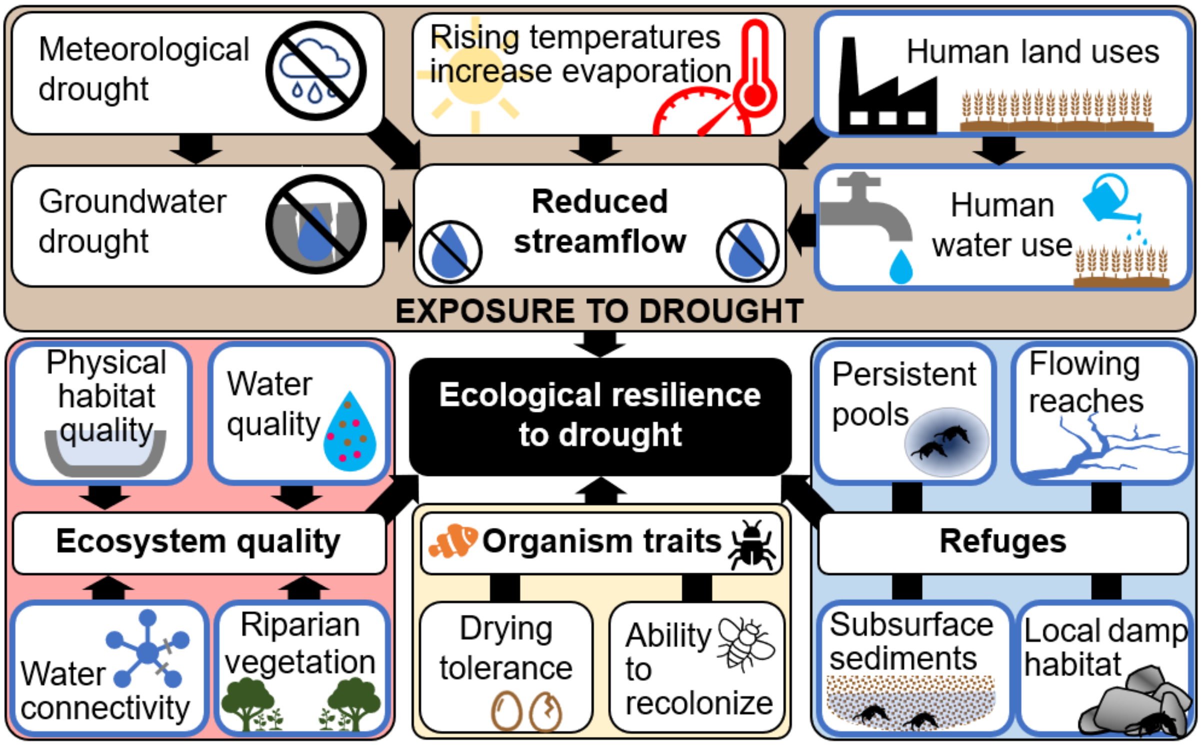


Figure 6. Factors affecting ecological resilience upon exposure to drought, where resilience is the capacity of biological communities in a river ecosystem to cope with the drought and recover after it ends. Exposure panel adapted from Crausbay et al. (2017); Refuges cartoons (including Gammarus, depicted [not to scale] as a representative invertebrate) adapted from Stubbington (2012). Thick blue borders indicate factors that managers can manipulate.

Take action: maintain flow. River managers must act to maintain sufficient flow to protect biodiversity within functional ecosystems. 'Sufficient' flow will differ among river types and among sites on any one river, with dry phases being natural, important components of temporary flow regimes, whereas perennial flow is ecologically desirable at historically perennial sites.

Sustainable drought management strategies do and must include hands-off flow conditions as strictly enforced abstraction licence constraints (Riley et al. 2009; Environment Agency 2021) that moderate human water use (Fig. 6). In addition, tailored restoration actions may be needed to protect key species that support ecosystem functioning. For example, restoration of physical habitats and thus flow regimes that include fast-flowing habitats can support natural recolonization of habitat-forming macrophytes such as Ranunculus (Fig. 6; Rangeley-Wilson 2021); alternatively, Ranunculus can establish after being transplanted (Riis et al. 2009), after which it (and other macrophytes) may slow the rate of drought-driven flow recession (O'Hare et al. 2010). In addition, fish of both socioeconomic and ecological importance such as Atlantic salmon Salmo salar should motivate significant investment in flow restoration to support their drought resilience (Beechie et al. 2013; Kastl et al. 2022). Such actions can have benefits beyond the target species.

Take action: maintain and enhance a diverse range of refuges. Dry phases represent particularly harsh environmental conditions for aquatic organisms, making wet habitats—including nearby flowing reaches; deep, persistent pools; saturated or moist, accessible subsurface sediments; and even localized damp algal mats—vital refuges in which organisms can survive and from which they can recolonize after flow resumes (Fig. 6; Chester and Robson 2011). Deeper or faster-flowing habitats also provide important low-flow refuges. The lack of such refuges increases drought impacts on communities in physically modified rivers (Dunbar et al. 2010a,b). As such, creating natural channel shapes including a diverse and extensive range of refuges should be central to restoration projects designed to enhance drought resilience, with Bond et al. (2008) arguing that such refuges "should be the highest priority for protection … during drought". The following steps could maintain and enhance refuges that promote drought resilience:

- **Identify and characterize existing refuges.** Network-scale management strategies should first identify where dry phases currently occur (as exemplified by monthly observations of wet and dry flow states in the Hertfordshire and North London Area; see Sefton et al. 2019) and use rainfall–runoff modelling to predict where it will occur in future (Sauquet et al. 2021). Then, for sites at which damaging dry phases or low flows are predicted, habitats which support biodiverse communities including target species could be mapped during the driest seasons in wet or 'normal' years (Tonkin et al. 2019). The results could identify existing high-quality habitats at sites with high refuge potential, guiding restoration actions elsewhere.

- **Promote connectivity to the hyporheic refuge.** The subsurface sediments of the hyporheic zone are a potentially extensive, persistent, local refuge that enables rapid recolonization after water returns (Hynes 1958; Vander Vorste et al. 2016a). However, fine sediment pollution routinely compromises in-channel habitat quality in English rivers (Naden et al. 2016), infiltrating coarser substrates and blocking the interstitial pathways that would otherwise enable migration into the hyporheic refuge (Boulton et al. 1998). Restoration of natural flow should therefore include actions both to monitor and manage diffuse silt pollution (Naura et al. 2016) and to increase connectivity with the subsurface sediments (Hester and Gooseff 2010), without which refuge-seeking migrations may not occur (Maazouzi et al. 2017).

- **Create new refuges**. Restoration of natural processes can maintain and enhance fast-flowing habitats and (introduced or naturally occurring) woody material, creating a diverse range of habitats including deep scour pools (Beechie et al. 2010; Kondolf et al. 2013). Such pools may act as refuges that sustain important fish populations as well as supporting wider drought resilience (Elliott 2000). Given the variable success of restoration projects, evidence-informed approaches should be developed, likely with input from the River Restoration Centre. Restoration of Australian and mediterranean-climate rivers to promote their drought resilience has had variable results, offering opportunities to learn from experience (Kondolf et al. 2013).

- **Promote connectivity to network-wide refuges.** High-quality refuges may not be used if they are inaccessible, and management actions should also seek to increase connectivity between refuges and drought-impacted sites (Fig. 6; Hughes 2007). Increasing connectivity could involve creation of terrestrial habitat corridors that support insect flight between streams (Robson et al. 2011) and prioritization of dam removals that enable fish to move to and from refuges (Gido et al. 2016).

- **Stock fish if necessary to avoid food web collapse.** Given the widespread occurrence of artificial barriers that limit fish migration through England's river network (Jones et al. 2019), stocking to replace lost fish populations may be required to avoid permanent shifts to simplified communities (Ledger et al. 2013). Human-facilitated reintroductions may also be required for species of conservation concern, as an interim measure pending restoration to support self-sustaining populations (e.g. Purse 2002).

Take action: promote natural drought management. The Rivers Trust is spearheading calls to "build back wetter": to incorporate features that help the environment store water introduced during wet periods and release it slowly, including via pathways that lead to stream channels (Cooper 2022). We need comparable investment in natural drought management as seen for natural flood management (Hankin et al. 2018); indeed, the two go hand-in-hand, with floodwater part of that which can be stored and released slowly—both to prevent flood peaks and to elongate flowing phases (Wohl et al. 2018). For example:

- Introducing (or allowing the natural accumulation of) large wood can hold up water in headwater streams, whilst also providing drought refuges for species including salmonids (Vehanen et al. 2010).

- Riparian tree planting can enhance ecosystem quality, providing future sources of large wood that promote natural hydrological variability and associated natural processes (Fig. 6). Riparian vegetation also increases shading, moderating water temperatures (Grabowski et al. 2019) and thus keeping coldwater species including brown trout within a tolerable thermal environment during summer drought (Broadmeadow et al. 2011). Planting may hasten reversal of drought-driven vegetation loss (Webb and Leake 2006); if so, selecting drought-tolerant species may—depending on how future climate change unfolds—promote their long-term persistence.

- Beaver introductions have considerable, if localized, potential to contribute to natural drought (and flood) management by creating dams that regulate downstream flow (Brazier et al. 2021).

Take action: drought-proof ecosystems by enhancing water quality. Throughout England's river networks, drought resilience could be enhanced by improving water quality and physical habitat quality (Fig. 6; Durance and Ormerod 2009). For example, Huml et al. (2019) suggest reducing phosphate, nitrate and metal pollution to promote resilience of grayling Thymallus thymallus to future climate change scenarios incorporating both temperature and river flow. Reducing phosphorus inputs can also mitigate the effects of phytoplankton blooms and filamentous algal growth during periods of low, slow flows and elevated water temperatures (Wilby et al. 1998; Bussi et al. 2016). During drought, it is also essential to prevent organic enrichment e.g. by sewage effluent, which can comprise ≤100% of discharge in streams that would otherwise be dry (Eppehimer et al. 2020) and can be poorly diluted elsewhere, reducing oxygen availability, in particular if temperatures are high (Durance and Ormerod 2009).

Use appropriate indices to inform timely action. The Environment Agency Drought Monitoring Network aims to enable timely identification of drought onset and its early ecological impacts, although its effectiveness will depend on the relative timing of drought onset, sample collection (in spring and autumn) and processing. The resultant macroinvertebrate community monitoring data are analysed using the LIFE (Lotic-invertebrate Index for Flow Evaluation) index (Extence et al. 1999) to determine long-term trends, with index values below certain thresholds triggering management actions (e.g. abstraction licence constraints) that mitigate drought impacts. LIFE has the advantage of its familiarity and integration into RIVPACS, enabling assessment of flow stress on macroinvertebrate communities by comparison of type-specific observed and expected index values (Clarke et al. 2003).

Additional macroinvertebrate-based biological indices could also prove informative indicators of the ecological impacts of drought. In particular, the DEHLI (Drought Effect of Habitat Loss on Invertebrates) index (Chadd et al. 2017) was developed to characterize community responses to drought-driven changes in habitat availability and may represent responses to drying more effectively than LIFE (Sarremejane et al. 2019). In addition, pressure-specific indices could be used to identify the ecological impacts of drought-associated stressors such as fine sediment (using the Proportion of Sediment-sensitive Invertebrates [PSI] index; Turley et al. 2015, 2016; Extence et al. 2017), salinity (using the Salinity Association Group Index [SAGI]; Pickwell et al. 2022) and organic enrichment (using WHPT; Paisley et al. 2014).

All such indices are designed to represent the community at a 'site', i.e. at the reach or sub-reach spatial scale. Drought has ecological effects that operate at spatial scales from the habitat patch to the catchment, making site-specific communities an appropriate, intermediate scale to characterize (Munson et al. 2021). In addition, such indices can be used to characterize ecological responses to drought at larger spatial scales including regional and national scales (e.g. Monk et al. 2008; Extence et al. 2017).

17. Research priorities: understanding drought in river ecosystems

Identify future tipping points

The risk of droughts that push ecosystems past tipping points to new stable states is ever increasing. As such, research priority #1 is to identify ecosystems on the approach to a tipping point, so that action can be taken to prevent catastrophic change. A decline in resilience (here, technically the capacity of an ecosystem to tolerate drought without shifting to a different stable state; Holling 1973) usually precedes a permanent state shift (Scheffer et al. 2001). Such declines can be identified using 'resilience indicators', providing early warning of a forthcoming state shift (Dakos 2015; van der Bolt et al. 2021). The length of the time series required to identify characteristic decline in the recovery of such indicators depends on the ecological response rate, enabling research in dynamic river ecosystems, in which most organisms complete their life cycles within weeks to a few years (van der Bolt et al. 2021).

Research is needed to characterize how resilience indicators respond to drought and to predict ecological responses to future scenarios, including increases: in drought frequency and associated cumulative effects of successive droughts; in drought duration, including multiyear events not seen in the UK in living memory; and in spatial extent, including drying of currently perennial reaches. Such modelling should consider interactions with other extreme events (e.g. floods and heatwaves) as well as recognizing the influence of other anthropogenic pressures (e.g. abstraction and pollution), as discussed below.

– **Application to management.** Modelling how resilience indicators respond to future drought scenarios will enable timely identification of rivers at risk of catastrophic change, prompting management actions that increase drought resilience and avoid state shifts.

Characterize interactions with other climatic extremes and other pressures

As long-lasting events, droughts are likely to coincide with other climatic extremes, including summer heatwaves, unseasonably mild winters, summer storms and winter floods. The unpredictability of such extremes means that we know little about how interactions between concurrent events (e.g. a drought and a heatwave) and successive events (e.g. consecutive dry winters) influence riverine communities and wider ecological resilience (but see e.g. Williams 2016). Second, riverine communities experience climate-driven drought alongside a range of human pressures, including abstraction, sewage pollution, agricultural and urban runoff and physical habitat modification. A meaningful, real-world understanding of drought responses needs to recognize these pressures, but their ubiquitous cooccurrence hampers characterization of the effects of individual stressors and of stressor combinations using field data.

Experimental research in outdoor mesocosms has enabled characterization of individual and interactive effects of 2–3 drought-related stressors (i.e. low flows, high temperatures, fine sediment and/or salinity) for periods of 1–2 months (e.g. Piggott et al. 2012; Beermann et al. 2018; Arias Font et al. 2021), but such valuable studies evidence the inevitable limitations of experimental work. Innovative analysis of long-term Environment Agency time-series data describing biological communities and environmental conditions (Sarremejane et al. 2019, 2020, 2021b) can complement experimental work to better represent both the range of stressors alongside which drought occurs as well as the timescales over which drought affects ecosystem structure and function.

– **Application to management.** Research that identifies alternative stressors (such as water quality) which can be manipulated to improve resilience to drought and other climate extremes could be crucial to supporting future river ecosystems.

Identify priority locations

Despite the goal of achieving good ecological status in all waterbodies, resources are limited and certain locations warrant prioritization. These locations could be highly impacted rivers that need salvaging, least-impacted rivers that need preserving to protect refuges that support recovery of other sites, or intermediate-quality sites with particular potential for improvement, for example, due to their network position.

Building on Monk et al. (2006), analysis of long-term Environment Agency time-series data describing environmental parameters and biological communities could enable identification and comparison of drought-sensitive and drought-resilient ‘natural’ river types, and how human impacts including physical habitat modification alter their drought resilience. Characterizing the features of identified drought-resilient sites could inform management actions that increase the capacity of drought-sensitive sites/communities to tolerate drought (Hain et al. 2018).

– **Application to management.** Identifying priority sites for action could ensure effective investment of limited resources in places that maximize local to catchment-scale drought resilience.

Research is needed to address biases in current evidence:

- **Spatial scale.** We know most about ecological responses to and recovery from drought at local, site-level scales, including within-site variability among habitat patches. We know less about network-scale responses, notwithstanding research in the Colne and Lee catchments, the results of which indicate variability among sites with contrasting flow regimes (Sarremejane et al. 2020, 2021b).

  Characterizing this spatial variability could inform catchment-scale management plans that support connected, drought-resilient metacommunities.

- **Biotic groups.** We know most about the responses of salmonids, benthic macroinvertebrates, macrophytes (including macroalgae) and biofilms, in particular diatoms. In contrast, most microorganisms (including bacteria and fungi, which are important decomposers) meiofauna (i.e. microscopic invertebrates), stygofauna (i.e. groundwater invertebrates), semi-aquatic taxa, riparian and terrestrial species (including river-dependent birds and mammals) are poorly studied.

  Characterizing these groups will create a holistic, catchment-scale understanding of ecological responses to drought that transcends boundaries between a river and adjacent ecosystems.

- **Insect adults.** Many riverine insect species have aquatic juvenile and terrestrial adult life stages, but almost all research has characterized how aquatic juveniles respond to drought. Considering the environmental requirements of both life stages could create a more complete understanding of species-specific responses to drought. For example, a soil moisture deficit that alters the structure of riparian vegetation may reduce survival and dispersal of terrestrial adult life stages (Robson et al. 2011; Larsen et al. 2016). Equally, slow flow velocities and dry phases may reduce the availability of waters upon which adult females can lay eggs (Hynes 1958; Iversen et al. 1978). In both cases, populations of aquatic juveniles may decline—even if their own habitat conditions are ideal.

- **Functional responses.** Most ecological studies characterize communities based on the taxa present and their individual and collective responses to environmental drivers. The concurrent responses of 'functional communities' (in which organisms are classified using their traits, such as their feeding mode) are less well characterized, and thus the consequences of shifts in taxonomic community composition for ecosystem functioning remain largely unknown (but see Ledger et al. 2012, 2013; Aspin et al. 2018, 2019a,b).

  Functional characterization could help to identify thresholds at which drought pushes ecosystems across thresholds at which their functioning is irreversibly impaired.

- **Ecosystem services.** Drought-driven changes to environmental conditions, biological communities and ecosystem functions will inevitably alter the benefits that rivers deliver to people, including cultural services such as recreation, provisioning services including water supply, and regulating services such as water purification (Lynch et al. 2023).

Characterizing how ecological responses alter ecosystem service delivery could promote holistic understanding of drought, motivating investment in management actions that enhance resilience of rivers as socioecological systems.

Glossary

Dry phase. I have minimized use of the word drying, which is technically a process, or transitional phase, in which surface water is lost from a stream. Instead, I use dry phase to refer to in-channel conditions in which surface water is either absent or restricted to isolated pools.

Ephemeral. Ephemeral flow permanence regimes are characterized by rain-driven, short-lived, unpredictable flowing phases and long dry phases, and hence 'terrestrialized' in-channel habitats. This definition is in line with core texts in the international discipline of temporary river science (Datry et al. 2017; Busch et al. 2020) and contrasts with the use of ephemeral in UK management contexts to refer to all streams which sometimes dry (which I term temporary, as defined below).

Deterministic extinction is an extinction caused by deterministic processes, i.e. extinction caused by the abiotic environment (e.g. a dry phase) and/or biotic interactions.

Extinction. I use this term loosely, to describe any complete loss of a species at a local or larger spatial scale. This local-scale use aligns with the technical term extirpation, definition of which recognizes that the species may subsequently recolonize.

Hyporheic zone. The hyporheic zone has sometimes been strictly defined as those subsurface sediments in which surface water and groundwater, and thus epigean and hypogean fauna, mix. However, I follow the broader, more inclusive definition of DelVecchia et al. (2022), which also recognizes hyporheic zones as occurring in temporary streams. Essentially, I use the term to refer to the subsurface sediments which are influenced by the surface stream but not exposed to much light or much flow.

Intermittence. Following Datry and Stubbington (2022), I define intermittence as the state or quality of being intermittent, ephemeral, or having any other temporary flow regime, i.e. one in which surface water sometimes stops flowing. As such, loss of surface flow (i.e. moving water) rather than loss of surface water (i.e. drying) is technically the defining feature of intermittence (Leigh et al. 2016). In practice, however, most authors (including me) use intermittence (and ephemeral, intermittent, non-perennial and temporary) to refer to streams that sometimes dry (i.e. lose most or all surface water, although pools of surface water may remain and subsurface sediments may be saturated).

Intermittent. Intermittent flow permanence regimes are characterized by predictable, groundwater-influenced, seasonal shifts between flowing, ponded and dry phases. For example, chalk winterbournes have intermittent flow regimes. This definition is in line with core texts in the international discipline of temporary river science (Datry et al. 2017; Busch et al. 2020). However, intermittent is often used to refer to all temporary flow regimes (e.g. England et al. 2019). Therefore, although redundant, I precede the word intermittent with seasonally, to ensure my meaning is clear.

Macrophyte. Macrophytes are submerged, emergent or floating aquatic plants including vascular flowering plants, mosses and liverworts and some encrusting lichens. Common definitions also consider a few large algae (such as filamentous green algae of the genus Cladophora) as 'macrophytes', but I avoid using the term in this sense.

Metacommunity. A metacommunity is a set of local (e.g. site-scale) communities that are linked by dispersal, as might occur in rivers and throughout catchments. Leibold et al. (2004) is the seminal paper on the metacommunity concept and Cid et al. (2020) consider metacommunity dynamics in a context of river management.

Metapopulation. A metapopulation is a set of local (e.g. site-scale) populations (i.e. of a single species) that are linked by dispersal (Gilpin and Hanski 1991), as might occur in rivers and throughout catchments

Near-perennial. Near-perennial flow regimes are characterized by occasional loss of surface flow and/or surface water, i.e. only during drought years.

Perennial. Perennial streams are those which never stop flowing, and which have never been known to dry.

Pressure. Variously defined in the literature, I follow recent Environment Agency (2019) reports in defining a pressure "as a factor affecting the water environment", and thus precede it with either human or anthropogenic (used interchangeably) where appropriate. This broad definition allows drought to be considered a pressure alongside broad factors such as urban land use and specific factors such as elevated nitrate concentrations—but I distinguish the latter specific factors as stressors (as defined below).

Richness. Taxonomic richness is the number of taxa per unit of measurement (e.g. an area of a streambed or a sample). Richness can be measured at any taxonomic level (e.g. order, family, genus or species), but is often (and in particular for aquatic macroinvertebrates) measured at a mixed family-to-species level.

River (cf. stream). I use river and stream somewhat interchangeably, but use river specifically when referring to larger watercourses.

Stochastic extinction is an extinction caused by demographic stochasticity (i.e. natural, random fluctuations in population densities among years) rather than environmental variation. Ovaskainen and Meerson (2010) provide more details.

Stream (cf. river). I use river and stream somewhat interchangeably, but use stream specifically when referring to smaller watercourses.

Stressor. Stressors refer to the specific natural and/or anthropogenic factors (such as elevated nitrate concentrations, or surface water loss) with potential to cause direct, physiological stress to organisms.

Temporary. Temporary is a comparable term to non-perennial and describes streams and flow regimes in which surface water sometimes stops flowing. As such, loss of surface flow (i.e. moving water) rather than loss of surface water (i.e. drying) is technically the defining feature of a temporary flow regime (Leigh et al. 2016). In practice, however, most authors (including me) use temporary to refer to streams that sometimes dry (i.e. lose most or all surface water, although pools of surface water may remain and subsurface sediments may be saturated). Temporary flow regimes include ephemeral, intermittent and near-perennial flow regimes.

Winterbourne. Winterbourne streams are intermittent streams that occur in the upper reaches of rivers fed by the chalk aquifer. Their surface flow originates from rising groundwater and springs that typically become active from December, with streamflow the continuing to rise until March/April before declining until winter (Berrie 1992). The length of the dry phase thus varies depending on a site's longitudinal position, making it difficult to state a typical duration (White et al. 2018).

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References

Aprahamian, M.W., Aprahamian, C.D., Baglinière, J.L., Sabatié, R. and Alexandrino, P. 2003. Alosa alosa and Alosa fallax spp. Literature review and bibliography. R&D Technical Report W1-014/TR. Environment Agency.

Archdeacon, T.P. and Reale, J.K., 2020. No quarter: Lack of refuge during flow intermittency results in catastrophic mortality of an imperiled minnow. Freshwater Biology 65: 2108–2123. https://doi.org/10.1111/fwb.13607

Argerich, A., Martí, E., Sabater, F., Ribot, M., von Schiller, D. and Riera, J.L., 2008. Combined effects of leaf litter inputs and a flood on nutrient retention in a Mediterranean mountain stream during fall. Limnology and Oceanography 53: 631–641. https://doi.org/10.4319/lo.2008.53.2.0631

Arias Font, R.A., Khamis, K., Milner, A.M., Smith, G.H.S. and Ledger, M.E., 2021. Low flow and heatwaves alter ecosystem functioning in a stream mesocosm experiment. Science of the Total Environment 777: 146067. https://doi.org/10.1016/j.scitotenv.2021.146067

Armitage, P.D. and Bass, J., 2013. Long-term resilience and short-term vulnerability of South Winterbourne macroinvertebrates. Proceedings of the Dorset Natural History and Archaeological Society 134: 43–55.

Armstrong, J.D., Braithwaite, V.A. and Fox, M., 1998. The response of wild Atlantic salmon parr to acute reductions in water flow. Journal of Animal Ecology 67: 292–297. https://doi.org/10.1046/j.1365-2656.1998.00193.x

Arnell, N.W., Halliday, S.J., Battarbee, R.W., Skeffington, R.A. and Wade, A.J., 2015. The implications of climate change for the water environment in England. Progress in Physical Geography 39: 93–120. https://doi.org/10.1177/0309133314560369

Aspin, T. and House, A., 2022. Alpha and beta diversity and species co-occurrence patterns in headwaters supporting rare intermittent-stream specialists. Freshwater Biology 67: 1188–1202. https://doi.org/10.1111/fwb.13910

Aspin, T.W., Matthews, T.J., Khamis, K., Milner, A.M., Wang, Z., O'Callaghan, M.J. and Ledger, M.E., 2018. Drought intensification drives turnover of structure and function in stream invertebrate communities. Ecography 41: 1992–2004. https://doi.org/10.1111/ecog.03711

Aspin, T.W., Hart, K., Khamis, K., Milner, A.M., O'Callaghan, M.J., Trimmer, M., Wang, Z., Williams, G.M., Woodward, G. and Ledger, M.E., 2019a. Drought intensification alters the composition, body size, and trophic structure of invertebrate assemblages in a stream mesocosm experiment. Freshwater Biology 64: 750–760. https://doi.org/10.1111/fwb.13259

Aspin, T.W., Khamis, K., Matthews, T.J., Milner, A.M., O'Callaghan, M.J., Trimmer, M., Woodward, G. and Ledger, M.E., 2019b. Extreme drought pushes stream invertebrate communities over functional thresholds. Global Change Biology 25: 230–244. https://doi.org/10.1111/gcb.14495

Attrill, M.J. and Power, M., 2000a. Modelling the effect of drought on estuarine water quality. Water Research 34: 1584–1594. https://doi.org/10.1016/S0043-1354(99)00305-X

Attrill, M.J. and Power, M., 2000b. Effects on invertebrate populations of drought-induced changes in estuarine water quality. Marine Ecology Progress Series 203: 133–143. https://doi.org/10.3354/meps203133

Attrill, M.J., Rundle, S.D. and Thomas R.M., 1996. The influence of drought-induced low freshwater flow on an upper-estuarine macroinvertebrate community. Water Research: 261–268. https://doi.org/10.1016/0043-1354(95)00186-7

Bachmair, S., Stahl, K., Collins, K., Hannaford, J., Acreman, M., Svoboda, M., Knutson, C., Smith, K.H., Wall, N., Fuchs, B. and Crossman, N.D., 2016. Drought indicators revisited: the need for a wider consideration of environment and society. Wiley Interdisciplinary Reviews: Water 3: 516–536. https://doi.org/10.1002/wat2.1154

Banha, F. and Anastácio, P.M., 2014. Desiccation survival capacities of two invasive crayfish species. Knowledge & Management of Aquatic Ecosystems 413: 01. https://doi.org/10.1051/kmae/2013084

Barker, L.J., Hannaford, J., Chiverton, A. and Svensson, C., 2016. From meteorological to hydrological drought using standardised indicators. Hydrology and Earth System Sciences 20: 2483–2505. https://doi.org/10.5194/hess-20-2483-2016

Barker, L.J., Hannaford, J., Parry, S., Smith, K.A., Tanguy, M. and Prudhomme, C., 2019. Historic hydrological droughts 1891–2015: systematic characterisation for a diverse set of catchments across the UK. Hydrology and Earth System Sciences 23: 4583–4602. https://doi.org/10.5194/hess-23-4583-2019

Bašić, T., Britton, J.R., Cove, R.J., Ibbotson, A.T. and Gregory, S.D., 2018. Roles of discharge and temperature in recruitment of a cold-water fish, the European grayling Thymallus thymallus, near its southern range limit. Ecology of Freshwater Fish 27: 940–951. https://doi.org/10.1111/eff.12405

Bass, J.A., Armitage, P.D. and Tapia, G. 2022. Aquatic invertebrates of Dorset winterbournes: a comparison with the small perennial chalk streams. Proceedings of the Dorset Natural History and Archaeological Society 143. https://www.dorsetmuseum.org/publications/dnhas-proceedings/

Beechie, T.J., Sear, D.A., Olden, J.D., Pess, G.R., Buffington, J.M., Moir, H., Roni, P. and Pollock, M.M., 2010. Process-based principles for restoring river ecosystems. BioScience 60: 209–222. https://doi.org/10.1525/bio.2010.60.3.7

Beechie, T., Imaki, H., Greene, J., Wade, A., Wu, H., Pess, G., Roni, P., Kimball, J., Stanford, J., Kiffney, P. and Mantua, N., 2013. Restoring salmon habitat for a changing climate. River Research and Applications 29: 939–960. https://doi.org/10.1002/rra.2590

Beerling, D.J., 1993. The impact of temperature on the northern distribution limits of the introduced species Fallopia japonica and Impatiens glandulifera in north-west Europe. Journal of Biogeography 20: 45–53. https://doi.org/10.2307/2845738

Beerling, D.J., Huntley, B. and Bailey, J.P., 1995. Climate and the distribution of Fallopia japonica: use of an introduced species to test the predictive capacity of response surfaces. Journal of Vegetation Science 6: 269–282. https://doi.org/10.2307/3236222

Beermann, A.J., Elbrecht, V., Karnatz, S., Ma, L., Matthaei, C.D., Piggott, J.J. and Leese, F., 2018. Multiple-stressor effects on stream macroinvertebrate communities: A mesocosm experiment manipulating salinity, fine sediment and flow velocity. Science of the Total Environment 610: 961–971. https://doi.org/10.1016/j.scitotenv.2017.08.084

Bennett, D.M., Dudley, T.L., Cooper, S.D. and Sweet, S.S., 2015. Ecology of the invasive New Zealand mud snail, Potamopyrgus antipodarum (Hydrobiidae), in a mediterranean-climate stream system. Hydrobiologia 746: 375–399. https://doi.org/10.1007/s10750-014-2136-6

Berrie, A.D., 1992. The chalk-stream environment. Hydrobiologia 248: 3–9. https://doi.org/10.1007/BF00008881

Bickerton, M.A., 1995. Long-term changes of macroinvertebrate communities in relation to flow variations: The river Glen, Lincolnshire, England. Regulated Rivers: Research & Management 10: 81–92. https://doi.org/10.1002/rrr.3450100204

Biondi, G., Gething, K.J., Kabir, J., Sykes, T. and Stubbington, R. 2022. The invertebrate biodiversity in the hidden headwater springs and streams of the southern Chalk. FBA News: the Freshwater Biological Association Newsletter 85: 8–11. Available at: http://irep.ntu.ac.uk/id/eprint/47092/1/1601100_Gething.pdf

Bogan, M.T., Chester, E.T., Datry, T., Murphy, A.L., Robson, B.J., Ruhi, A., Stubbington, R. and Whitney, J.E., 2017. Resistance, resilience, and community recovery in intermittent rivers and ephemeral streams, pp. 349–376 in: Datry, T., Bonada, N. and Boulton, A.J. (Eds.). Intermittent Rivers and Ephemeral Streams. Academic Press. https://doi.org/10.1016/B978-0-12-803835-2.00013-9

Bond, N.R., Lake, P.S. and Arthington, A.H., 2008. The impacts of drought on freshwater ecosystems: an Australian perspective. Hydrobiologia 600: 3–16. https://doi.org/10.1007/s10750-008-9326-z

Boulton, A.J., 2003. Parallels and contrasts in the effects of drought on stream macroinvertebrate assemblages. Freshwater Biology 48: 1173–1185. https://doi.org/10.1046/j.1365-2427.2003.01084.x

Boulton, A.J. and Lake, P.S., 2008. Effects of drought on stream insects and its ecological consequences, pp. 81–102 in: Lancaster, J. and Briers, R.A. (Eds.). Aquatic Insects. Challenges to Populations. CAB International.

Boulton, A.J., Findlay, S., Marmonier, P., Stanley, E.H. and Valett, H.M., 1998. The functional significance of the hyporheic zone in streams and rivers. Annual Review of Ecology and Systematics 29: 59–81. http://www.jstor.org/stable/221702

Bowes, M.J., Loewenthal, M., Read, D.S., Hutchins, M.G., Prudhomme, C., Armstrong, L.K., Harman, S.A., Wickham, H.D., Gozzard, E. and Carvalho, L., 2016. Identifying multiple stressor controls on phytoplankton dynamics in the River Thames (UK) using high-frequency water quality data. Science of the Total Environment 569: 1489–1499. https://doi.org/10.1016/j.scitotenv.2016.06.239

Bowes, M., O'Brien, A., Hutchins, M. and Laize, C., 2023. Thematic review 3: Water quality. Environment Agency report series: Review of the state of research on drought.

Brazier, R.E., Puttock, A., Graham, H.A., Auster, R.E., Davies, K.H. and Brown, C.M., 2021. Beaver: Nature's ecosystem engineers. Wiley Interdisciplinary Reviews: Water 8: e1494. https://doi.org/10.1002/wat2.1494

Brittain, J.E. and Eikeland, T.J., 1988 Invertebrate drift — A review. Hydrobiologia 166: 77–93. https://doi.org/10.1007/BF00017485

Broadmeadow, S.B., Jones, J.G., Langford, T.E.L., Shaw, P.J. and Nisbet, T.R., 2011. The influence of riparian shade on lowland stream water temperatures in southern England and their viability for brown trout. River Research and Applications 27: 226–237. https://doi.org/10.1002/rra.1354

Brooker, M.P., Morris, D.L. and Hemsworth, R.J., 1977. Mass mortalities of adult salmon, Salmo salar, in the R. Wye, 1976. Journal of Applied Ecology 14: 409–417. https://doi.org/10.2307/2402554

Buisson, L., Thuiller, W., Lek, S., Lim, P. and Grenouillet, G., 2008. Climate change hastens the turnover of stream fish assemblages. Global Change Biology 14: 2232–2248. https://doi.org/10.1111/j.1365-2486.2008.01657.x

Buisson, L., Grenouillet, G., Villéger, S., Canal, J. and Laffaille, P., 2013. Toward a loss of functional diversity in stream fish assemblages under climate change. Global Change Biology 19: 387–400. https://doi.org/10.1111/gcb.12056

Busch, M.H., Costigan, K.H., Fritz, K.M., Datry, T., Krabbenhoft, C.A., Hammond, J.C., Zimmer, M., Olden, J.D., Burrows, R.M., Dodds, W.K. and Boersma, K.S., 2020. What's in a name? Patterns, trends, and suggestions for defining non-perennial rivers and streams. Water 12: 1980. https://doi.org/10.3390/w12071980

Bussi, G., Whitehead, P.G., Bowes, M.J., Read, D.S., Prudhomme, C. and Dadson, S.J., 2016. Impacts of climate change, land-use change and phosphorus reduction on phytoplankton in the River Thames (UK). Science of the Total Environment 572: 1507–1519. https://doi.org/10.1016/j.scitotenv.2016.02.109

Calapez, A.R., Elias, C.L., Almeida, S.F. and Feio, M.J., 2014. Extreme drought effects and recovery patterns in the benthic communities of temperate streams. Limnetica 33: 281–296. https://doi.org/10.23818/limn.33.22

Cañedo-Argüelles, M., Bogan, M.T., Lytle, D.A. and Prat, N., 2016. Are Chironomidae (Diptera) good indicators of water scarcity? Dryland streams as a case study. Ecological Indicators 71 155–162. https://doi.org/10.1016/j.ecolind.2016.07.002

Caramujo, M.J., Mendes, C.R.B., Cartaxana, P., Brotas, V. and Boavida, M.J., 2008. Influence of drought on algal biofilms and meiofaunal assemblages of temperate reservoirs and rivers. Hydrobiologia 598 77–94. https://doi.org/10.1007/s10750-007-9141-y

Chadd, R.P., England, J.A., Constable, D., Dunbar, M.J., Extence, C.A., Leeming, D.J., Murray-Bligh, J.A. and Wood, P.J., 2017. An index to track the ecological effects of drought development and recovery on riverine invertebrate communities. Ecological Indicators 82: 344–356. https://doi.org/10.1016/j.ecolind.2017.06.058

Chergui, H. and Pattee, E., 1988. The impact of benthic invertebrates on the breakdown of poplar leaves in the network of a large European river. Archiv für Hydrobiologie 113: 15–25. https://doi.org/10.1127/archiv-hydrobiol/113/1989/15

Chester, E.T. and Robson, B.J., 2011. Drought refuges, spatial scale and recolonisation by invertebrates in non-perennial streams. Freshwater Biology 56: 2094–2104. https://doi.org/10.1111/j.1365-2427.2011.02644.x

Cid, N., Bonada, N., Heino, J., Cañedo-Argüelles, M., Crabot, J., Sarremejane, R., Soininen, J., Stubbington, R. and Datry, T., 2020. A metacommunity approach to improve biological assessments in highly dynamic freshwater ecosystems. Bioscience 70: 427–438. https://doi.org/10.1093/biosci/biaa033

Colls, M., Timoner, X., Font, C., Sabater, S. and Acuña, V., 2019. Effects of duration, frequency, and severity of the non-flow period on stream biofilm metabolism. Ecosystems 22: 1393–1405. https://doi.org/10.1007/s10021-019-00345-1

Cole, G.A. and Marsh, T.J., 2006. The impact of climate change on severe droughts: Major droughts in England and Wales from 1800 and evidence of impact: Science Report: SC040068/SR1. Environment Agency.

Cooper, E., 2022. Drought is our 'new normal' — we need to build back wetter for the long term [online]. Available at: https://theriverstrust.org/about-us/news/drought-announcement

Copp, G.H. and Vilizzi, L., 2004. Spatial and ontogenetic variability in the microhabitat use of stream-dwelling spined loach (Cobitis taenia) and stone loach (Barbatula barbatula). Journal of Applied Ichthyology 20: 440–451. https://doi.org/10.1111/j.1439-0426.2004.00605.x

Corrarino, C.A. and Brusven, M.A., 1983. The effects of reduced stream discharge on insect drift and stranding of near shore insects. Freshwater Invertebrate Biology 2: 88–98. https://doi.org/10.2307/1467113

Cosgrove, P., Shields, D., Anderson, D. and Massey, K., 2021. Understanding the impact of the 2018 drought on key freshwater pearl mussel populations in Scotland. NatureScot Research Report No. 1210.

Cotton, J.A., Wharton, G., Bass, J.A.B., Heppell, C.M. and Wotton, R.S., 2006. The effects of seasonal changes to in-stream vegetation cover on patterns of flow and accumulation of sediment. Geomorphology 77: 320–334. https://doi.org/10.1016/j.geomorph.2006.01.010

Cowx, I.G., Young, W.O. and Hellawell, J.M., 1984. The influence of drought on the fish and invertebrate populations of an upland stream in Wales. Freshwater Biology 14: 165–177. https://doi.org/10.1111/j.1365-2427.1984.tb00030.x

Crabot, J., Polášek, M., Launay, B., Pařil, P. and Datry, T., 2021. Drying in newly intermittent rivers leads to higher variability of invertebrate communities. Freshwater Biology 66: 730–744. https://doi.org/10.1111/fwb.13673

Cranston, E. and Darby, E., 2002. Ranunculus in chalk rivers, phase 2. R&D Technical Report W1-042/TR. Environment Agency.

Crausbay, S.D., Ramirez, A.R., Carter, S.L., Cross, M.S., Hall, K.R., Bathke, D.J., Betancourt, J.L., Colt, S., Cravens, A.E., Dalton, M.S. and Dunham, J.B., 2017. Defining ecological drought for the twenty-first century. Bulletin of the American Meteorological Society 98: 2543–2550. https://doi.org/10.1175/BAMS-D-16-0292.1

Crausbay, S.D., Betancourt, J., Bradford, J., Cartwright, J., Dennison, W.C., Dunham, J., Enquist, C.A., Frazier, A.G., Hall, K.R., Littell, J.S. and Luce, C.H., 2020. Unfamiliar territory: Emerging themes for ecological drought research and management. One Earth 3: 337–353. https://doi.org/10.1016/j.oneear.2020.08.019

Culling, M.A., Lozano Valero, I. and Côté, I.M., 2003. Substratum preferences and diel activity patterns of spined loach Cobitis taenia in England: implications for conservation management. Folia Biologica (Krakow) 51 (Suppl.): 129–133.

Curley, E.A., Thomas, R., Adams, C.E. and Stephen, A., 2022. Adaptive responses of freshwater pearl mussels, Margaritifera margaritifera, to managed drawdowns. Aquatic Conservation: Marine and Freshwater Ecosystems 32: 466–483. https://doi.org/10.1002/aqc.3759

Daguet, C.A., French, G.C. and Taylor, P. (Eds.). 2008. The Odonata Red Data List for Great Britain – species status no. 11. Joint Nature Conservation Committee. ISSN 1473-0154. Available via: https://hub.jncc.gov.uk/assets/8e18c30d-794e-4b0c-8d1c-378c3faf0e7c

Dakos, V., Carpenter, S.R., van Nes, E.H. and Scheffer, M., 2015. Resilience indicators: prospects and limitations for early warnings of regime shifts. Philosophical Transactions of the Royal Society B: Biological Sciences 370: 20130263. https://doi.org/10.1098/rstb.2013.0263

Datry T. and Stubbington, R., 2022. Intermittent rivers and ephemeral streams, pp. 628–640 in: Tockner, K. (Ed.) Encyclopedia of Inland Waters 2nd ed. vol. 2. Elsevier. https://doi.org/10.1016/B978-0-12-819166-8.00090-6

Datry, T., Corti, R., Claret, C. and Philippe, M., 2011. Flow intermittence controls leaf litter breakdown in a French temporary alluvial river: the “drying memory”. Aquatic Sciences 73: 471–483. https://doi.org/10.1007/s00027-011-0193-8

Datry, T., Bonada, N. and Boulton, A., 2017. Intermittent rivers and ephemeral streams: ecology and management, pp. 1–20 in: Datry, T., Bonada, N. and Boulton, A.J. (Eds.). Intermittent Rivers and Ephemeral Streams. Academic Press. https://doi.org/10.1016/B978-0-12-803835-2.00001-2

Debowski, P. and Beall, E., 1995. Influence of dewatering on movements and distribution of salmon parr (Salmo salar L.) in relation to habitat characteristics in an experimental stream. Bulletin Français de la Pêche et de la Pisciculture 337–339 : 267–275. https://dx.doi.org/10.1051/kmae:1995030

Defra (Department for Environment, Food & Rural Affairs), 2013. Managing abstraction and the water environment. Defra.

Defra (Department for Environment, Food & Rural Affairs), 2022. Guidance: Habitats and species of principal importance in England. Defra. Available at: https://www.gov.uk/government/publications/habitats-and-species-of-principal-importance-in-england

DelVecchia, A.G., Shanafield, M., Zimmer, M.A., Busch, M.H., Krabbenhoft, C.A., Stubbington, R., Kaiser, K.E., Burrows, R.M., Hosen, J., Datry, T. and Kampf, S.K., 2022. Reconceptualizing the hyporheic zone for nonperennial rivers and streams. Freshwater Science 41: 167–182. https://doi.org/10.1086/720071

Dewson, Z.S., James, A.B.W. and Death, R.G., 2007. A review of the consequences of decreased flow for instream habitat and macroinvertebrates. Journal of the North American Benthological Society 26: 401–415. https://doi.org/10.1899/06-110.1

Diamond, J.S., Moatar, F., Cohen, M.J., Poirel, A., Martinet, C., Maire, A. and Pinay, G., 2022. Metabolic regime shifts and ecosystem state changes are decoupled in a large river. Limnology and Oceanography 67: S54–S70. https://doi.org/10.1002/lno.11789

Dobel, A.J., O'Hare, M., Gunn, I. and Edwards, F., 2020. Streams and rivers: Report card 2020. About Drought. Available at: https://aboutdrought.info/wp-content/uploads/2020/07/AboutDrought-ReportCard-Rivers-Streams-Final.pdf

Dollar, E. et al., 2013. Monitoring and assessment of environmental impacts of droughts: Literature synthesis: Report SC120024/R1. Environment Agency. Available at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/291497/LIT_8569_b765dd.pdf

Dorts, J., Grenouillet, G., Douxfils, J., Mandiki, S.N., Milla, S., Silvestre, F. and Kestemont, P., 2012. Evidence that elevated water temperature affects the reproductive physiology of the European bullhead Cottus gobio. Fish Physiology and Biochemistry 38: 389–399. https://doi.org/10.1007/s10695-011-9515-y

Drake, C.M., 2017. A review of the status of larger Brachycera flies of Great Britain – species status no. 29. Natural England Commissioned Report NECR192. Natural England. Available at: http://publications.naturalengland.org.uk/file/5193416897462272

Driver, L.J. and Hoeinghaus, D.J., 2016. Fish metacommunity responses to experimental drought are determined by habitat heterogeneity and connectivity. Freshwater Biology 61: 533–548. https://doi.org/10.1111/fwb.12726

Drummond, L.R., McIntosh, A.R. and Larned, S.T., 2015. Invertebrate community dynamics and insect emergence in response to pool drying in a temporary river. Freshwater Biology 60: 1596–1612. https://doi.org/10.1111/fwb.12591

Dunbar, M.J., Pedersen, M.L., Cadman, D., Extence, C., Waddingham, J., Chadd, R. and Larsen, S.E., 2010a. River discharge and local-scale physical habitat influence macroinvertebrate LIFE scores. Freshwater Biology 55: 226–242. https://doi.org/10.1111/j.1365-2427.2009.02306.x

Dunbar, M.J., Warren, M., Extence, C., Baker, L., Cadman, D., Mould, D.J., Hall, J. and Chadd, R., 2010b. Interaction between macroinvertebrates, discharge and physical habitat in upland rivers. Aquatic Conservation: Marine and Freshwater Ecosystems 20: S31–S44. https://doi.org/10.1002/aqc.1089

Durance, I. and Ormerod, S.J. 2009. Trends in water quality and discharge confound long-term warming effects on river macroinvertebrates. Freshwater Biology 54: 388–405. https://doi.org/10.1111/j.1365-2427.2008.02112.x

Durance, I. and Ormerod, S.J., 2010. Evidence for the role of climate in the local extinction of a cool-water triclad. Journal of the North American Benthological Society 29: 1367–1378. https://doi.org/10.1899/09-159.1

Durkota, J.M., Wood, P.J., Johns, T., Thompson, J.R. and Flower, R.J., 2019. Distribution of macroinvertebrate communities across surface and groundwater habitats in response to hydrological variability. Fundamental and Applied Limnology 193: 79–92. https://doi.org/10.1127/fal/2019/1156

Ellerby, D.J., Spierts, I.L.Y. and Altringham, J.D., 2001. Fast muscle function in the European eel (Anguilla anguilla L.) during aquatic and terrestrial locomotion. Journal of Experimental Biology 204: 2231–2238. https://doi.org/10.1242/jeb.204.13.2231

Elliott, J.M., 2000. Pools as refugia for brown trout during two summer droughts: trout responses to thermal and oxygen stress. Journal of Fish Biology 56: 938–948. https://doi.org/10.1111/j.1095-8649.2000.tb00883.x

Elliott, J.M., 2006. Periodic habitat loss alters the competitive coexistence between brown trout and bullheads in a small stream over 34 years. Journal of Animal Ecology 75: 54–63. https://doi.org/10.1111/j.1365-2656.2005.01022.x

Ellis, A. and England, J.A. 2009. Crayfish in Hertfordshire [online]. Available to ResearchGate members.

England, J., Chadd, R., Dunbar, M.J., Sarremejane, R., Stubbington, R., Westwood, C.G. and Leeming, D., 2019. An invertebrate-based index to characterize ecological responses to flow intermittence in rivers. Fundamental and Applied Limnology 193: 93–117. https://doi.org/10.1127/fal/2019/1206

Environment Agency, 2019. Phosphorus and freshwater eutrophication pressure narrative. Environment Agency.

Environment Agency, 2021. Policy paper: Managing water abstraction. Environment Agency. Available at: https://www.gov.uk/government/publications/managing-water-abstraction/managing-water-abstraction

Eppehimer, D. E., Hamdhani, H., Hollien, K. D. and Bogan, M. T. 2020. Evaluating the potential of treated effluent as novel habitats for aquatic invertebrates in arid regions. Hydrobiologia 847: 3381–3396. https://doi.org/10.1007/s10750-020-04343-6

Everall, N.C., Johnson, M.F., Wilby, R.L. and Bennett, C.J., 2015. Detecting phenology change in the mayfly Ephemera danica: responses to spatial and temporal water temperature variations. Ecological Entomology 40: 95–105. https://doi.org/10.1111/een.12164

Everard, M., 1996. The importance of periodic droughts for maintaining diversity in the freshwater environment. Freshwater Forum 7: 33–50. Available at: http://hdl.handle.net/1834/22179

Extence, C.A., 1981. The effect of drought on benthic invertebrate communities in a lowland river. Hydrobiologia 83: 217–224. https://doi.org/10.1007/BF00008269

Extence, C.A., Balbi, D.M. and Chadd, R.P., 1999. River flow indexing using British benthic macroinvertebrates: a framework for setting hydroecological objectives. Regulated Rivers: Research & Management 15: 545–574. https://doi.org/10.1002/(SICI)1099-1646(199911/12)15:6%3C545::AID-RRR561%3E3.0.CO;2-W

Extence, C.A., Chadd, R.P., England, J., Naura, M. and Pickwell, A.G.G., 2017. Application of the proportion of sediment-sensitive invertebrates (PSI) biomonitoring index. River Research and Applications 33: 1596–1605. https://doi.org/10.1002/rra.3227

Fausch, K.D., Torgersen, C.E., Baxter, C.V. and Li, H.W., 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes: a continuous view of the river is needed to understand how processes interacting among scales set the context for stream fishes and their habitat. BioScience 52: 483–498. https://doi.org/10.1641/0006-3568(2002)052[0483:LTRBTG]2.0.CO;2

Filbee-Dexter, K., Pittman, J., Haig, H.A., Alexander, S.M., Symons, C.C. and Burke, M.J., 2017. Ecological surprise: concept, synthesis, and social dimensions. Ecosphere 8: e02005. https://doi.org/10.1002/ecs2.2005

Fleig, A.K., Tallaksen, L.M., Hisdal, H. and Demuth, S., 2006. A global evaluation of streamflow drought characteristics. Hydrology and Earth System Sciences 10: 535–552. https://doi.org/10.5194/hess-10-535-2006

Flynn, N.J., Snook, D.L., Wade, A.J. and Jarvie, H.P., 2002. Macrophyte and periphyton dynamics in a UK Cretaceous chalk stream: the River Kennet, a tributary of the Thames.

Science of the Total Environment 282: 143–157. https://doi.org/10.1016/S0048-9697(01)00949-4

Fonseca, C.R. and Ganade, G., 2001. Species functional redundancy, random extinctions and the stability of ecosystems. Journal of Ecology 89: 118–125. https://doi.org/10.1046/j.1365-2745.2001.00528.x

Foster, G.N., 2010. A review of the scarce and threatened Coleoptera of Great Britain Part (3): Water beetles of Great Britain – species status no. 1. Joint Nature Conservation Committee. Available at: https://data.jncc.gov.uk/data/e80beaf2-456c-4178-8f91-16c2ef01368d/SpeciesStatus-1-Coleoptera-WEB-2010.pdf

Fournier, R.J., de Mendoza, G., Sarremejane, R. and Ruhi, A., 2023. Isolation controls reestablishment mechanisms and post-drying community structure in an intermittent stream. Ecology 104: e3911. https://doi.org/10.1002/ecy.3911

Fowler, K., Peel, M., Saft, M., Nathan, R., Horne, A., Wilby, R., McCutcheon, C. and Peterson, T., 2022. Hydrological shifts threaten water resources. Water Resources Research 58: e2021WR031210. https://doi.org/10.1029/2021WR031210

Franklin, P., Dunbar, M. and Whitehead, P., 2008. Flow controls on lowland river macrophytes: a review. Science of the Total Environment 400: 369–378. https://doi.org/10.1016/j.scitotenv.2008.06.018

Freshwater Biological Association, 1977–79. Ecological study of chalk streams. Lambourn project. Interim report October 1977 to April 1979. Freshwater Biological Association. Available at: http://hdl.handle.net/1834/22617

Galen, C., 2005. It never rains but then it pours: the diverse effects of water on flower integrity and function, pp. 77–95 in: Reekie, E.G. and Bazzaz, F.A. (Eds.). Reproductive Allocation in Plants. Academic Press. https://doi.org/10.1016/B978-012088386-8/50003-X

Garssen, A.G., Verhoeven, J.T. and Soons, M.B., 2014. Effects of climate-induced increases in summer drought on riparian plant species: A meta-analysis. Freshwater Biology 59: 1052–1063. https://doi.org/10.1111/fwb.12328

Gessner, M.O., Chauvet, E. and Dobson, M., 1999. A perspective on leaf litter breakdown in streams. Oikos 85: 377–384. https://doi.org/10.2307/3546505

Gido, K.B., Whitney, J.E., Perkin, J.S. and Turner, T.F., 2016. Fragmentation, connectivity and fish species persistence in freshwater ecosystems, pp. 292–323 in: Closs, G.P., Krkosek, M. and Olden, J.D. (Eds.). Conservation of Freshwater Fishes. Cambridge University Press. Available to ResearchGate members.

Giling, D., Reich, P. and Thompson, R.M., 2009. Loss of riparian vegetation alters the ecosystem role of a freshwater crayfish (Cherax destructor) in an Australian intermittent lowland stream. Journal of the North American Benthological Society 28: 626–637. https://doi.org/10.1899/09-015.1

Gilpin, M.E. and Hanski, I.A., 1991. Metapopulation Dynamics: Empirical and Theoretical Investigations. Academic Press.

Gómez, R., Arce, M.I., Baldwin, D.S. and Dahm, C.N., 2017. Water physicochemistry in intermittent rivers and ephemeral streams, pp. 109–134 in: Datry, T., Bonada, N. and Boulton, A.J. (Eds.). Intermittent Rivers and Ephemeral Streams. Academic Press. https://doi.org/10.1016/B978-0-12-803835-2.00005-X

Grabowski, R.C., Gurnell, A.M., Burgess-Gamble, L., England, J., Holland, D., Klaar, M.J., Morrissey, I., Uttley, C. and Wharton, G., 2019. The current state of the use of large wood in river restoration and management. Water and Environment Journal 33: 366–377. https://doi.org/10.1111/wej.12465

Gurnell, A.M., Van Oosterhout, M.P., De Vlieger, B. and Goodson, J.M., 2006. Reach-scale interactions between aquatic plants and physical habitat: River Frome, Dorset. River Research and Applications 22: 667–680. https://doi.org/10.1002/rra.929

Gurnell, A.M., Scott, S.J., England, J., Gurnell, D., Jeffries, R., Shuker, L. and Wharton, G., 2020. Assessing river condition: A multiscale approach designed for operational application in the context of biodiversity net gain. River Research and Applications 36: 1559–1578. https://doi.org/10.1002/rra.3673

Hain, E.F., Kennen, J.G., Caldwell, P.V., Nelson, S.A., Sun, G. and McNulty, S.G., 2018. Using regional scale flow–ecology modeling to identify catchments where fish assemblages are most vulnerable to changes in water availability. Freshwater Biology 63: 928–945. https://doi.org/10.1111/fwb.13048

Hall, J.W., Hannaford, J. and Hegerl, G., 2022. Drought risk in the Anthropocene. Philosophical Transactions of the Royal Society A 380: 20210297. https://doi.org/10.1098/rsta.2021.0297

Ham, S.F., Wright, J.F. and Berrie, A.D., 1981. Growth and recession of aquatic macrophytes on an unshaded section of the River Lambourn, England, from 1971 to 1976. Freshwater Biology 11: 381–390. https://doi.org/10.1111/j.1365-2427.1981.tb01269x

Hankin, B., Chappell, W., Page, T., Kipling, K., Whitling, M. and Burgess-Gamble, L. 2018. Environment Agency. Mapping the potential for working with natural processes. Technical Report SC150005.

Hannaford, J., 2015. Climate-driven changes in UK river flows: A review of the evidence. Progress in Physical Geography 39: 29–48. https://doi.org/10.1177/0309133314536755

Hannaford, J. and Buys, G. 2012. Seasonal trends in river flow regimes and extremes in the UK. Journal of Hydrology 475: 158–174. http://dx.doi.org/10.1016/j.jhydrol.2012.09.044

Hannaford, J., Collins, K., Haines, S. and Barker, L.J., 2019. Enhancing drought monitoring and early warning for the United Kingdom through stakeholder coinquiries. Weather, Climate, and Society 11: 49–63. https://doi.org/10.1175/WCAS-D-18-0042.1

Hannaford, J., Mackay, J., Ascott, M., Bell, V., Chitson, T., Cole, S., Counsell, C., Durant, M., Jackson, C.R., Kay, A. and Lane, R., 2022. eFLaG: enhanced future Flows and Groundwater. A national dataset of hydrological projections based on UKCP18. Earth System Science Data Discussions [preprint]. https://doi.org/10.5194/essd-2022-40

Hannaford, J., Barker, L., Turner, S., Parry, S., Tanguy, M. and Chevuturi, A., 2023. Past variability in observed hydrological (river flow) drought. Environment Agency report series: Review of the state of research on drought.

Hannah, D., Khamis, K. and White, J. 2023. Reviewing river temperature dynamics during droughts across the United Kingdom. Environment Agency report series: Review of the state of research on drought.

Hayes, C., 2022. Ecological responses to environmental variability in wet and dry chalk streams: towards better temporary stream monitoring and management. PhD thesis. Nottingham Trent University. Available at: https://irep.ntu.ac.uk/id/eprint/48573/

Heffernan, J.B., 2008. Wetlands as an alternative stable state in desert streams. Ecology 89: 1261–1271. https://doi.org/10.1890/07-0915.1

Hester, E.T. and Gooseff, M.N., 2010. Moving beyond the banks: Hyporheic restoration is fundamental to restoring ecological services and functions of streams. Environmental Science & Technology 44: 1521–1525. https://doi.org/10.1021/es902988n

Hill, M.J. and Milner, V.S., 2018. Ponding in intermittent streams: A refuge for lotic taxa and a habitat for newly colonising taxa? Science of the Total Environment 628: 1308–1316. https://doi.org/10.1016/j.scitotenv.2018.02.162

Hill, M.J., Mathers, K.L., Little, S., Worrall, T., Gunn, J. and Wood, P.J., 2019. Ecological effects of a supra-seasonal drought on macroinvertebrate communities differ between near-perennial and ephemeral river reaches. Aquatic Sciences 81: 62. https://doi.org/10.1007/s00027-019-0659-7

Holdich, D.M., James, J., Jackson, C. and Peay, S., 2014. The North American signal crayfish, with particular reference to its success as an invasive species in Great Britain. Ethology Ecology & Evolution 26: 232–262. https://doi.org/10.1080/03949370.2014.903380

Holling, C.S., 1973. Resilience and stability of ecological systems. Annual Review of Ecology and Systematics 4: 1–23. https://www.jstor.org/stable/2096802

Holmes, N.T.H., 1999. Recovery of headwater stream flora following the 1989-1992 groundwater

drought. Hydrological Processes 13: 341–354. https://doi.org/10.1002/(SICI)1099-1085(19990228)13:3%3C341::AID-HYP742%3E3.0.CO;2-L

House, A., Punchard, N. and Bowles, F. 2008. Fish out of water- use of the River Avon winterbournes. Institute of Fisheries Management conference 2008. Unpublished.

Hughes, J.M., 2007. Constraints on recovery: using molecular methods to study connectivity of aquatic biota in rivers and streams. Freshwater Biology 52: 616–631. https://doi.org/10.1111/j.1365-2427.2006.01722.x

Hulme, P.E., 2017. Climate change and biological invasions: evidence, expectations, and response options. Biological Reviews 92: 1297–1313. https://doi.org/10.1111/brv.12282

Huml, J.V., Harris, W.E., Taylor, M.I., Sen, R., Prudhomme, C. and Ellis, J.S., 2020. Pollution control can help mitigate future climate change impact on European grayling in the UK. Diversity and Distributions 26: 17–532. https://doi.org/10.1111/ddi.13039

Hundecha, Y. and Bárdossy, A., 2004. Modeling of the effect of land use changes on the runoff generation of a river basin through parameter regionalization of a watershed model. Journal of Hydrology 292: 281–295. https://doi.org/10.1016/j.jhydrol.2004.01.002

Hynes, H.B.N., 1958. The effect of drought on the fauna of a small mountain stream in Wales. Internationale Vereinigung für Theoretische und Angewandte Limnologie: Verhandlungen 13: 826–833. https://doi.org/10.1080/03680770.1956.11895473

IPCC (Intergovernmental Panel on Climate Change), 2022. Climate change 2022: Impacts, adaptation and vulnerability. Contribution of Working Group II to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. 3056 pp. Cambridge University Press. https://report.ipcc.ch/ar6/wg2/IPCC_AR6_WGII_FullReport.pdf

Iversen, T.M., Wiberg-Larsen, P., Hansen, S.B. and Hansen, F.S., 1978. The effect of partial and total drought on the macroinvertebrate communities of three small Danish streams. Hydrobiologia 60: 235–242. https://doi.org/10.1007/BF00011718

Jaeger, K.L., Olden, J.D. and Pelland, N.A., 2014. Climate change poised to threaten hydrologic connectivity and endemic fishes in dryland streams. Proceedings of the National Academy of Sciences 111: 13894–13899. https://doi.org/10.1073/pnas.1320890111

Jansson, R., Laudon, H., Johansson, E. and Augspurger, C., 2007. The importance of groundwater discharge for plant species number in riparian zones. Ecology 88: 131–139. https://doi.org/10.1890/0012-9658(2007)88[131:TIOGDF]2.0.CO;2

Jarvie, H.P., Neal, C., Warwick, A., White, J., Neal, M., Wickham, H.D., Hill, L.K. and Andrews, M.C., 2002. Phosphorus uptake into algal biofilms in a lowland chalk river. Science of the Total Environment 282: 353–373. https://doi.org/10.1016/S0048-9697(01)00924-X

JNCC (Joint Nature Conservation Committee), 2022a. 1103 Twaite shad Alosa fallax [online]. JNCC. Available at: https://sac.jncc.gov.uk/species/S1103/

JNCC (Joint Nature Conservation Committee), 2022b. Conservation designations for UK taxa – collation [online]. JNCC. Available at: https://jncc.gov.uk/our-work/conservation-designations-for-uk-taxa/

JNCC (Joint Nature Conservation Committee), 2022c. 3260 Water courses of plain to montane levels with the Ranunculion fluitantis and Callitricho-Batrachion vegetation [online]. JNCC. Available at: https://sac.jncc.gov.uk/habitat/H3260/

Johns, T., Smith, D.C., Homann, S. and England, J.A., 2018. Time-series analysis of a native and a non-native amphipod shrimp in two English rivers. BioInvasions Record 7: 101–110. https://doi.org/10.3391/bir.2018.7.2.01

Jones, J.I., Collins, A.L., Naden, P.S. and Sear, D.A., 2012a. The relationship between fine sediment and macrophytes in rivers. River Research and Applications 28: 1006–1018. https://doi.org/10.1002/rra.1486

Jones, J.I., Murphy, J.F., Collins, A.L., Sear, D.A., Naden, P.S. and Armitage, P.D., 2012. The impact of fine sediment on macro-invertebrates. River Research and Applications 28: 1055–1071. https://doi.org/10.1002/rra.1516

Jones, P.D. and Lister, D.H., 1998. Riverflow reconstructions for 15 catchments over England and Wales and an assessment of hydrologic drought since 1865. International Journal of Climatology: A Journal of the Royal Meteorological Society 18: 999–1013. https://doi.org/10.1002/(SICI)1097-0088(199807)18:9%3C999::AID-JOC300%3E3.0.CO;2-8

Jones, J., Börger, L., Tummers, J., Jones, P., Lucas, M., Kerr, J., Kemp, P., Bizzi, S., Consuegra, S., Marcello, L. and Vowles, A., 2019. A comprehensive assessment of stream fragmentation in Great Britain. Science of the Total Environment 673: 756–762. https://doi.org/10.1016/j.scitotenv.2019.04.125

Jonsson, B. and Jonsson, N., 2009. A review of the likely effects of climate change on anadromous Atlantic salmon Salmo salar and brown trout Salmo trutta, with particular reference to water temperature and flow. Journal of Fish Biology 75: 2381–2447. https://doi.org/10.1111/j.1095-8649.2009.02380.x

Jonsson, B., Jonsson, N. and Hansen, L.P., 2007. Factors affecting river entry of adult Atlantic salmon in a small river. Journal of Fish Biology 71: 943–956. https://doi.org/10.1111/j.1095-8649.2007.01555.x

Kastl, B., Obedzinski, M., Carlson, S.M., Boucher, W.T. and Grantham, T.E. 2022. Migration in drought: Receding streams contract the seaward migration window of endangered salmon. Ecosphere 13: e4295. https://doi.org/10.1002/ecs2.4295

Kawanishi, R., Inoue, M., Dohi, R., Fujii, A. and Miyake, Y., 2013. The role of the hyporheic zone for a benthic fish in an intermittent river: a refuge, not a graveyard. Aquatic Sciences 75: 425–431. https://doi.org/10.1007/s00027-013-0289-4

Kelly, M., 2014a. A “red list” of endangered British diatoms? [online]. Available at: https://microscopesandmonsters.wordpress.com/2014/11/16/a-red-list-of-endangered-british-diatoms/

Kelly, M., 2014b. The Really Rare Diatom Show [online]. Available at: https://microscopesandmonsters.wordpress.com/2014/11/18/the-really-rare-diatom-show/

Kelly, D.W., Dick, J.T. and Montgomery, W.I., 2002. The functional role of Gammarus (Crustacea, Amphipoda): shredders, predators, or both? Hydrobiologia 485: 199–203. https://doi.org/10.1023/A:1021370405349

Kemp, P., Sear, D., Collins, A., Naden, P. and Jones, I., 2011. The impacts of fine sediment on riverine fish. Hydrological Processes 25: 1800–1821. https://doi.org/10.1002/hyp.7940

Kendon, M., Marsh, T. and Parry, S., 2013. The 2010–2012 drought in England and Wales. Weather 68: 88–95. https://doi.org/10.1002/wea.2101

Kerezsy, A., Gido, K., Magalhães, M.F. and Skelton, P.H., 2017. The biota of intermittent rivers and ephemeral streams: Fishes, pp. 273–298 in: Datry, T., Bonada, N. and Boulton, A.J. (Eds.). Intermittent Rivers and Ephemeral Streams. Academic Press. https://doi.org/10.1016/B978-0-12-803835-2.00010-3

Keyantash, J.A. and Dracup, J.A., 2004. An aggregate drought index: Assessing drought severity based on fluctuations in the hydrologic cycle and surface water storage. Water Resources Research 40. https://doi.org/10.1029/2003WR002610

Kirby, K.J., Reid, C.M., Thomas, R.C. and Goldsmith, F.B., 1998. Preliminary estimates of fallen dead wood and standing dead trees in managed and unmanaged forests in Britain. Journal of Applied Ecology 35: 148–155. https://doi.org/10.1046/j.1365-2664.1998.00276.x

Kitchen, L., Macadam, C.R. and Yeomans, W.E., 2010. Is the upland summer mayfly (Ameletus inopinatus Eaton 1887) in hot water? Freshwater Biological Association et al.

Klaus, J., Monk, W.A., Zhang, L. and Hannah, D.M., 2022. Ecohydrological interactions during drought. Ecohydrology 15: e2456. https://doi.org/10.1002/eco.2456

Kondolf, G.M., Podolak, K. and Grantham, T.E., 2013. Restoring mediterranean-climate rivers. Hydrobiologia 719: 527–545. https://doi.org/10.1007/s10750-012-1363-y

Ladle, M. and Bass, J.A.B., 1981. The ecology of a small chalk stream and its responses to drying during drought conditions. Archiv für Hydrobiologie 90: 448–466

Ladle, M., Bass, J.A.B. and Jenkins, W.R., 1972. Studies on production and food consumption by the larval Simuliidae (Diptera) of a chalk stream. Hydrobiologia 39: 429–448. https://doi.org/10.1007/BF00046654

Lancaster, J. and Ledger, M.E., 2015. Population-level responses of stream macroinvertebrates to drying can be density-independent or density-dependent. Freshwater Biology 60: 2559–2570. https://doi.org/10.1111/fwb.12643

Lane, R. and Kay, A., 2023. Modelling climate change impacts on UK hydrological drought: a review. Environment Agency report series: Review of the state of research on drought.

Langhans, S.D. and Tockner, K., 2014. Edge effects are important in supporting beetle biodiversity in a gravel-bed river floodplain. PloS One 9: e114415. https://doi.org/10.1016/j.gecco.2014.08.009

Larsen, S., Muehlbauer, J.D. and Marti, E., 2016. Resource subsidies between stream and terrestrial ecosystems under global change. Global Change Biology 22: 2489–2504. https://doi.org/10.1111/gcb.13182

Lassalle, G. and Rochard, E., 2009. Impact of twenty-first century climate change on diadromous fish spread over Europe, North Africa and the Middle East. Global Change Biology 15: 1072–1089. https://doi.org/10.1111/j.1365-2486.2008.01794.x

Lawson, D.A. and Rands, S.A., 2019. The effects of rainfall on plant–pollinator interactions. Arthropod-Plant Interactions 13: 561–569. https://doi.org/10.1007/s11829-019-09686-z

Leberfinger, K., Bohman, I. and Herrmann, J. 2010. Drought impact on stream detritivores: Experimental effects on leaf litter breakdown and life cycles. Hydrobiologia 652: 247–254. https://doi.org/10.1007/s10750-010-0337-1

Ledger, M.E. and Hildrew, A.G., 2001. Recolonization by the benthos of an acid stream following a drought. Archiv für Hydrobiologie 152: 1–17. https://doi.org/10.1127/archiv-hydrobiol/152/2001/1

Ledger, M.E., Harris, R.M., Armitage, P.D. and Milner, A.M., 2008. Disturbance frequency influences patch dynamics in stream benthic algal communities. Oecologia 155: 809–819. https://doi.org/10.1007/s00442-007-0950-5

Ledger, M.E., Edwards, F.K., Brown, L.E., Milner, A.M. and Woodward, G., 2011. Impact of simulated drought on ecosystem biomass production: an experimental test in stream mesocosms. Global Change Biology 17: 2288–2297. https://doi.org/10.1111/j.1365-2486.2011.02420.x

Ledger, M.E., Harris, R.M., Armitage, P.D. and Milner, A.M., 2012. Climate change impacts on community resilience: evidence from a drought disturbance experiment. Advances in Ecological Research 46: 211–258. https://doi.org/10.1016/B978-0-12-396992-7.00003-4

Ledger, M.E., Brown, L.E., Edwards, F.K., Milner, A.M. and Woodward, G., 2013. Drought alters the structure and functioning of complex food webs. Nature Climate Change 3: 223–227. https://doi.org/10.1038/nclimate1684

Leibold, M.A., Holyoak, M., Mouquet, N., Amarasekare, P., Chase, J.M., Hoopes, M.F., Holt, R.D., Shurin, J.B., Law, R., Tilman, D. and Loreau, M., 2004. The metacommunity concept: a framework for multi-scale community ecology. Ecology Letters 7: 601-613. https://doi.org/10.1111/j.1461-0248.2004.00608.x

Leigh, C., Boulton, A.J., Courtwright, J.L., Fritz, K., May, C.L., Walker, R.H. and Datry, T., 2016. Ecological research and management of intermittent rivers: an historical review and future directions. Freshwater Biology 61: 1181–1199. https://doi.org/10.1111/fwb.12646

Leigh, C., Aspin, T.W., Matthews, T.J., Rolls, R.J. and Ledger, M.E., 2019. Drought alters the functional stability of stream invertebrate communities through time. Journal of Biogeography 46: 1988–2000. https://doi.org/10.1111/jbi.13638

Loskotová, B., Straka, M. and Pařil, P., 2019. Sediment characteristics influence benthic macroinvertebrate vertical migrations and survival under experimental water loss conditions. Fundamental and Applied Limnology 193: 39–49. https://doi.org/10.1127/fal/2019/1138

Lu, X., Gray, C., Brown, L.E., Ledger, M.E., Milner, A.M., Mondragón, R.J., Woodward, G. and Ma, A., 2016. Drought rewires the cores of food webs. Nature Climate Change 6: 875–878. https://doi.org/10.1038/nclimate3002

Lynch, A.J. et al. 2023. People need freshwater biodiversity. Wiley Interdisciplinary Reviews: Water e1663. https://doi.org/10.1002/wat2.1633

Lytle, D.A. and Poff, N.L., 2004. Adaptation to natural flow regimes. Trends in Ecology & Evolution 19: 94–100. https://doi.org/10.1016/j.tree.2003.10.002

Maazouzi, C., Galassi, D., Claret, C., Cellot, B., Fiers, F., Martin, D., Marmonier, P. and Dole-Olivier, M.J., 2017. Do benthic invertebrates use hyporheic refuges during streambed drying? A manipulative field experiment in nested hyporheic flowpaths. Ecohydrology 10: e1865. https://doi.org/10.1002/eco.1865

Macadam, C.R., 2015. A review of the stoneflies (Plecoptera) of Great Britain – species status no. 20. Natural England Commissioned Report NECR174. Natural England. Available at: http://publications.naturalengland.org.uk/publication/5404611030548480

Macadam, C.R., 2016. A review of the mayflies (Ephemeroptera) of Great Britain – species status no. 28. Natural England Commissioned Report NECR193. Natural England. Available at: http://publications.naturalengland.org.uk/publication/4635857668538368

Macadam, C., Stubbington, R. and Wallace, I., 2021. The specialist insects that rely on the wet-dry habitats of temporary streams. FBA News: the Freshwater Biological Association Newsletter 81: 28–33. Available at: http://irep.ntu.ac.uk/id/eprint/44944/1/1498769_Stubbington.pdf

Macadam, C.R., England, J. and Chadd, R., 2022. The vulnerability of British aquatic insects to climate change. Knowledge & Management of Aquatic Ecosystems 423: 3. https://doi.org/10.1051/kmae/2022003

Macneil, C., Dick, J.T. and Elwood, R.W., 1999. The dynamics of predation on Gammarus spp.(Crustacea: Amphipoda). Biological Reviews 74: 375–395. https://doi.org/10.1017/S0006323199005368

Mann, R.H.K., 1989. Fish population dynamics in the River Frome, Dorset. Regulated Rivers: Research & Management 4: 165–177. https://doi.org/10.1002/rrr.3450040209

Mann, R.H.K. and Blackburn, J.H., 1991. The biology of the eel Anguilla anguilla (L.) in an English chalk stream and interactions with juvenile trout Salmo trutta L. and salmon Salmo salar L. Hydrobiologia 218: 65–76. https://doi.org/10.1007/BF00006419

Mann, R.H.K., Blackburn, J.H. and Beaumont, W.R.C., 1989. The ecology of brown trout Salmo trutta in English chalk streams. Freshwater Biology 21: 57–70. https://doi.org/10.1111/j.1365-2427.1989.tb01348.x

Marsh, T., Cole, G. and Wilby, R., 2007. Major droughts in England and Wales, 1800–2006. Weather 62: 87–93. https://doi.org/10.1002/wea.67

Marsh, J.E., Cove, R.J., Britton, J.R., Wellard, R.G., House, A. and Gregory, S.D., 2021. Medium-term environmental changes influence age-specific survival estimates in a salmonid population. Freshwater Biology 66: 1530–1545. https://doi.org/10.1111/fwb.13736

Masefield, S., 2018. Overland movement and activity of the endangered white-clawed crayfish Austropotamobius Pallipe. MRes thesis, University of Gloucestershire. Available via: https://eprints.glos.ac.uk/7059/

Mathers, K.L., White, J.C., Fornaroli, R. and Chadd, R., 2020. Flow regimes control the establishment of invasive crayfish and alter their effects on lotic macroinvertebrate communities. Journal of Applied Ecology 57: 886–902. https://doi.org/10.1111/1365-2664.13584

Matthews, W.J. and Marsh-Matthews, E., 2003. Effects of drought on fish across axes of space, time and ecological complexity. Freshwater Biology 48: 1232–1253. https://doi.org/10.1046/j.1365-2427.2003.01087.x

McIntosh, A., 2019. Vulnerability of freshwater ecosystems to tipping points. New Zealand's Biological Heritage: National Science Challenges. Available at: https://bioheritage.nz/wp-content/uploads/2019/04/2019-FEB-Tipping-Points-policy-brief-McIntosh.pdf

Met Office, 2013. England and Wales drought 2010 to 2012 [online]. Met Office.

Milly, P.C., Betancourt, J., Falkenmark, M., Hirsch, R.M., Kundzewicz, Z.W., Lettenmaier, D.P. and Stouffer, R.J., 2008. Stationarity is dead: Whither water management? Science 319: 573–574. https://doi.org/10.1126/science.1151915

Minaudo, C., Abonyi, A., Leitão, M., Lançon, A.M., Floury, M., Descy, J.P. and Moatar, F., 2021. Long-term impacts of nutrient control, climate change, and invasive clams on phytoplankton and cyanobacteria biomass in a large temperate river. Science of the Total Environment 756: 144074. https://doi.org/10.1016/j.scitotenv.2020.144074

Mitchell, S.C. and Cunjak, R.A., 2007. Relationship of upstream migrating adult Atlantic salmon (Salmo salar) and stream discharge within Catamaran Brook, New Brunswick. Canadian Journal of Fisheries and Aquatic Sciences 64: 563–573. https://doi.org/10.1139/f07-032

Monk, W.A., Wood, P.J., Hannah, D.M., Wilson, D.A., Extence, C.A. and Chadd, R.P., 2006. Flow variability and macroinvertebrate community response within riverine systems. River Research and Applications 22: 595–615. https://doi.org/10.1002/rra.933

Monk, W.A., Wood, P.J., Hannah, D.M. and Wilson, D.A., 2008. Macroinvertebrate community response to inter-annual and regional river flow regime dynamics. River Research and Applications 24: 988–1001. https://doi.org/10.1002/rra.1120

Moon, H.P., 1956. Observations on a small portion of a drying chalk stream. Proceedings of the Zoological Society of London 126: 327–334. https://doi.org/10.1111/j.1096-3642.1956.tb00441.x

Moore, J.W., 1977. Seasonal succession of algae in a eutrophic stream in southern England. Hydrobiologia 53: 181–192. https://doi.org/10.1007/BF00029297

Morrison, B.R.S., 1990. Recolonisation of four small streams in central Scotland following drought conditions in 1984. Hydrobiologia 208: 261–267. https://doi.org/10.1007/BF00007791

Mosley, L.M., 2015. Drought impacts on the water quality of freshwater systems; review and integration. Earth-Science Reviews 140: 203–214. https://doi.org/10.1016/j.earscirev.2014.11.010

Moss, B., 2014. Fresh waters, climate change and UK nature conservation. Freshwater Reviews 7: 25–75. https://doi.org/10.1608/FRJ-7.1.789

Munson, S.M., Bradford, J.B. and Hultine, K.R., 2021. An integrative ecological drought framework to span plant stress to ecosystem transformation. Ecosystems 24: 739–754. https://doi.org/10.1007/s10021-020-00555-y

Naden, P.S., Murphy, J.F., Old, G.H., Newman, J., Scarlett, P., Harman, M., Duerdoth, C.P., Hawczak, A., Pretty, J.L., Arnold, A. and Laizé, C., 2016. Understanding the controls on deposited fine sediment in the streams of agricultural catchments. Science of the Total Environment 547: 366–381. https://doi.org/10.1016/j.scitotenv.2015.12.079

Naura, M., Hornby, D.D., Collins, A.L., Sear, D.A., Hill, C., Jones, J.I. and Naden, P.S., 2016. Mapping the combined risk of agricultural fine sediment input and accumulation for riverine ecosystems across England and Wales. Ecological Indicators 70: 209–221. https://doi.org/10.1016/j.ecolind.2016.03.055

NBN (National Biodiversity Network), 2021. Pseudanodonta complanata (Rossmässler, 1835). Depressed River Mussel. NBN Atlas Partnership. Available at: https://species.nbnatlas.org/species/NHMSYS0000336867#overview

Nelson, D., Busch, M.H., Kopp, D.A. and Allen, D.C., 2021. Energy pathways modulate the resilience of stream invertebrate communities to drought. Journal of Animal Ecology 90: 2053–2064. https://doi.org/10.1111/1365-2656.13490

NIDIS (National Integrated Drought Information System), 2023. What is drought: Ecological drought [online]. NIDIS. Available at: https://www.drought.gov/what-is-drought/ecological-drought

Nogueira, J.G., Lopes-Lima, M., Varandas, S., Teixeira, A. and Sousa, R., 2021. Effects of an extreme drought on the endangered pearl mussel Margaritifera margaritifera: a before/after assessment. Hydrobiologia 848: 3003–3013. https://doi.org/10.1007/s10750-019-04103-1

Nunn, A.D., Copp, G.H., Vilizzi, L. and Carter, M.G., 2010. Seasonal and diel patterns in the migrations of fishes between a river and a floodplain tributary. Ecology of Freshwater Fish 19: 153–162. https://doi.org/10.1111/j.1600-0633.2009.00399.x

O'Hare, M.T., McGahey, C., Bissett, N., Cailes, C., Henville, P. and Scarlett, P., 2010. Variability in roughness measurements for vegetated rivers near base flow, in England and Scotland. Journal of Hydrology 385: 361–370. https://doi.org/10.1016/j.jhydrol.2010.02.036

Ovaskainen, O. and Meerson, B., 2010. Stochastic models of population extinction. Trends in Ecology & Evolution 25: 643–652. https://doi.org/10.1016/j.tree.2010.07.009

Paisley, M.F., Trigg, D.J. and Walley, W.J., 2014. Revision of the biological monitoring working party (BMWP) score system: Derivation of present-only and abundance-related scores from field data. River Research and Applications 30: 887–904. https://doi.org/10.1002/rra.2686

Pařil, P., Leigh, C., Polášek, M., Sarremejane, R., Řezníčková, P., Dostálová, A. and Stubbington, R., 2019. Short-term streambed drying events alter amphipod population structure in a central European stream. Fundamental and Applied Limnology 193: 51–64. http://doi.org/10.1127/fal/2019/1164

Parmesan, C., 2006. Ecological and evolutionary responses to recent climate change. Annual Review of Ecology, Evolution, and Systematics 37: 637–669. https://doi.org/10.1146/annurev.ecolsys.37.091305.110100

Parr, L.B. and Mason, C.F., 2003. Long-term trends in water quality and their impact on macroinvertebrate assemblages in eutrophic lowland rivers. Water Research 37: 2969–2979. https://doi.org/10.1016/S0043-1354(03)00115-5

Parry, S., Prudhomme, C., Wilby, R.L. and Wood, P.J., 2016a. Drought termination: Concept and characterisation. Progress in Physical Geography 40: 743–767. https://doi.org/10.1177/0309133316652801

Parry, S., Wilby, R.L., Prudhomme, C. and Wood, P.J., 2016b. A systematic assessment of drought termination in the United Kingdom. Hydrology and Earth System Sciences 20: 4265–4281. https://doi.org/10.5194/hess-20-4265-2016

Patel, C., Vadher, A.N., Mathers, K.L., Dwyer, C. and Wood, P.J., 2021. Body size affects the vertical movement of benthic amphipods through subsurface sediments in response to drying. Hydrobiologia 848: 1015–1025. https://doi.org/10.1007/s10750-020-04500-x

Pendergrass, A.G. et al., 2020. Flash droughts present a new challenge for subseasonal-to-seasonal prediction. Nature Climate Change 10: 191–199. https://doi.org/10.1038/s41558-020-0709-0

Perkin, J.S., Gido, K.B., Costigan, K.H., Daniels, M.D. and Johnson, E.R., 2015. Fragmentation and drying ratchet down Great Plains stream fish diversity. Aquatic Conservation: Marine and Freshwater Ecosystems 25: 639–655. https://doi.org/10.1002/aqc.2501

Perkin, J.S., Starks, T.A., Pennock, C.A., Gido, K.B., Hopper, G.W. and Hedden, S.C., 2019. Extreme drought causes fish recruitment failure in a fragmented Great Plains riverscape. Ecohydrology 12: e2120. https://doi.org/10.1002/eco.2120

Perrow, M., Leeming, D., England, J. and Tomlinson, M., 2007. Life after low flow – ecological recovery of the River Misbourne. British Wildlife 18: 335–346.

Peterson, T., Saft, M., Peel, M.C. and John, A., 2021. Watersheds may not recover from drought. Science 372: 745–749. https://doi.org/10.1126/science.abd5085

Pickwell, A., Constable, D., Chadd, R., Extence, C. and Little, S., 2022. The development of a novel macroinvertebrate indexing tool for the determination of salinity effects in freshwater habitats. River Research and Applications 38: 522–538. https://doi.org/10.1002/rra.3914

Piggott, J.J., Lange, K., Townsend, C.R. and Matthaei, C.D., 2012. Multiple stressors in agricultural streams: a mesocosm study of interactions among raised water temperature, sediment addition and nutrient enrichment. PloS One 7: e49873. https://doi.org/10.1371/journal.pone.0049873

Pinder, A.C., Hopkins, E., Scott, L.J. and Britton, J.R., 2016. Rapid visual assessment of spawning activity and associated habitat utilisation of sea lamprey (Petromyzon marinus Linnaeus, 1758) in a chalk stream: implications for conservation monitoring. Journal of Applied Ichthyology 32: 364–368. https://doi.org/10.1111/jai.13010

Poff, N.L., Allan, J.D., Bain, M.B., Karr, J.R., Prestegaard, K.L., Richter, B.D., Sparks, R.E. and Stromberg, J.C., 1997. The natural flow regime. BioScience 47: 769–784. https://doi.org/10.2307/1313099

Power, M.E., Tilman, D., Estes, J.A., Menge, B.A., Bond, W.J., Mills, L.S., Daily, G., Castilla, J.C., Lubchenco, J. and Paine, R.T., 1996. Challenges in the quest for keystones: identifying keystone species is difficult—but essential to understanding how loss of species will affect ecosystems. BioScience 46: 609–620. https://doi.org/10.2307/1312990

Poznańska, M., Kakareko, T., Gulanicz, T., Jermacz, Ł. and Kobak, J., 2015. Life on the edge: survival and behavioural responses of freshwater gill-breathing snails to declining water level and substratum drying. Freshwater Biology 60: 2379–2391. https://doi.org/10.1111/fwb.12664

Prenda, J., Armitage, P.D. and Grayston, A., 1997. Habitat use by the fish assemblages of two chalk streams. Journal of Fish Biology 51: 64–79. https://doi.org/10.1111/j.1095-8649.1997.tb02514.x

Purse, B., 2002. The Ecology and Conservation of the Southern Damselfly (Coenagrion mercuriale – Charpentier) in Britain. R&D Technical Report W1-021/TR. Environment Agency. Available at: https://eprints.soton.ac.uk/436444/1/Purse_2002.pdf

Rangeley-Wilson, C., 2021. Chalk stream restoration strategy 2021: Main report [online]. Catchment Based Approach. Available at: https://catchmentbasedapproach.org/wp-content/uploads/2021/10/CaBA-CSRG-Strategy-MAIN-REPORT-FINAL-12.10.21-Low-Res.pdf

Reich, P., Lake, P.S., Thomson, J.R., Daniel, T., Johnson, M., Metzeling, L., Boulton, A.J. and Hale, R., 2023. Aquatic invertebrate responses to riparian restoration and flow extremes in three degraded intermittent streams: An eight-year field experiment. Freshwater Biology 68: 325–339. https://doi.org/10.1111/fwb.14028

Reynoldson, T., 1953. Habitat of Polycelis felina (= cornuta) and Crenobia alpina in the British Isles. Nature 171: 660. https://doi.org/10.1038/171660a0

Ribot, M., Grimm, N.B., Pollard, L.D., von Schiller, D., Handler, A.M. and Martí, E., 2022. Consequences of an ecosystem state shift for nitrogen cycling in a desert stream. Limnology and Oceanography 67: 1274–1286. https://doi.org/10.1002/lno.12076

Riis, T., Schultz, R., Olsen, H.M. and Katborg, C.K., 2009. Transplanting macrophytes to rehabilitate streams: experience and recommendations. Aquatic Ecology 43: 935. https://doi.org/10.1007/s10452-008-9213-8

Riley, W.D., Ives, M.J., Pawson, M.G. and Maxwell, D.L., 2006. Seasonal variation in habitat use by salmon, Salmo salar, trout, Salmo trutta and grayling, Thymallus thymallus, in a chalk stream. Fisheries Management and Ecology 13: 221–236. https://doi.org/10.1111/j.1365-2400.2006.00496.x

Riley, W.D., Maxwell, D.L., Pawson, M.G. and Ives, M.J., 2009. The effects of low summer flow on wild salmon (Salmo salar), trout (Salmo trutta) and grayling (Thymallus thymallus) in a small stream. Freshwater Biology 54: 2581–2599. https://doi.org/10.1111/j.1365-2427.2009.02268.x

Riley, W.D., Walker, A.M., Bendall, B. and Ives, M.J., 2011. Movements of the European eel (Anguilla anguilla) in a chalk stream. Ecology of Freshwater Fish 20: 628–635. https://doi.org/10.1111/j.1600-0633.2011.00513.x

Robotham, P.W.J., 1978. Some factors influencing the microdistribution of a population of spined loach, Cobitis taenia (L.). Hydrobiologia 61: 161–167. https://doi.org/10.1007/BF00018747

Robinson, R.A., Crick, H.Q., Learmonth, J.A., Maclean, I.M., Thomas, C.D., Bairlein, F., Forchhammer, M.C., Francis, C.M., Gill, J.A., Godley, B.J. and Harwood, J., 2009. Travelling through a warming world: climate change and migratory species. Endangered Species Research 7: 87–99. https://doi.org/10.3354/esr00095

Robson, B.J., Chester, E.T. and Austin, C.M., 2011. Why life history information matters: drought refuges and macroinvertebrate persistence in non-perennial streams subject to a drier climate. Marine and Freshwater Research 62: 801–810. https://doi.org/10.1071/MF10062

Rodríguez-Lozano, P., Leidy, R.A. and Carlson, S.M., 2019. Brook lamprey survival in the dry riverbed of an intermittent stream. Journal of Arid Environments 166: 83–85. https://doi.org/10.1016/j.jaridenv.2019.04.016

Romaní, A.M., Amalfitano, S., Artigas, J., Fazi, S., Sabater, S., Timoner, X., Ylla, I. and Zoppini, A., 2013. Microbial biofilm structure and organic matter use in mediterranean streams. Hydrobiologia 719: 43–58. https://doi.org/10.1007/s10750-012-1302-y

Romaní, A.M., Chauvet, E., Febria, C., Mora-Gómez, J., Risse-Buhl, U., Timoner, X., Weitere, M. and Zeglin, L., 2017. The biota of intermittent rivers and ephemeral streams: prokaryotes, fungi, and protozoans, pp. 161–188 in: Datry, T., Bonada, N. and Boulton, A.J. (Eds.). Intermittent Rivers and Ephemeral Streams. Academic Press. https://doi.org/10.1016/B978-0-12-803835-2.00009-7

Ruiz-Navarro, A., Gillingham, P.K. and Britton, J.R., 2016. Predicting shifts in the climate space of freshwater fishes in Great Britain due to climate change. Biological Conservation 203: 33–42. https://doi.org/10.1016/j.biocon.2016.08.021

Rudd, A.C., Kay, A.L. and Bell, V.A., 2019. National-scale analysis of future river flow and soil moisture droughts: potential changes in drought characteristics. Climatic Change 156: 323–340. https://doi.org/10.1007/s10584-019-02528-0

Ryo, M., Aguilar-Trigueros, C.A., Pinek, L., Muller, L.A. and Rillig, M.C., 2019. Basic principles of temporal dynamics. Trends in Ecology & Evolution 34: 723–733. https://doi.org/10.1016/j.tree.2019.03.007

Sabater, S., Timoner, X., Bornette, G., De Wilde, M., Stromberg, J.C. and Stella, J.C., 2017. The biota of intermittent rivers and ephemeral streams: Algae and vascular plants, pp. 189–216 in: Datry, T., Bonada, N. and Boulton, A.J. (Eds.). Intermittent Rivers and Ephemeral Streams. Academic Press. https://doi.org/10.1016/B978-0-12-803835-2.00016-4

Samaniego, L., Thober, S., Kumar, R., Wanders, N., Rakovec, O., Pan, M., Zink, M., Sheffield, J., Wood, E.F. and Marx, A., 2018. Anthropogenic warming exacerbates European soil moisture droughts. Nature Climate Change 8: 421–426. https://doi.org/10.1038/s41558-018-0138-5

Saltveit, S.J. and Brabrand, Å., 2013. Incubation, hatching and survival of eggs of Atlantic salmon (Salmo salar) in spawning redds influenced by groundwater. Limnologica 43: 325–331. https://doi.org/10.1016/j.limno.2013.05.009

Sarremejane, R., Mykrä, H., Huttunen, K.L., Mustonen, K.R., Marttila, H., Paavola, R., Sippel, K., Veijalainen, N. and Muotka, T., 2018. Climate-driven hydrological variability determines inter-annual changes in stream invertebrate community assembly. Oikos 127: 1586–1595. https://doi.org/10.1111/oik.05329

Sarremejane, R., Stubbington, R., Dunbar, M.J., Westwood, C.G. and England, J., 2019. Biological indices to characterize community responses to drying in streams with contrasting flow permanence regimes. Ecological Indicators: 107 105620. https://doi.org/10.1016/j.ecolind.2019.105620

Sarremejane, R., England, J., Sefton, C.E., Parry, S., Eastman, M. and Stubbington, R., 2020. Local and regional drivers influence how aquatic community diversity, resistance and resilience vary in response to drying. Oikos 129: 1877–1890. https://doi.org/10.1111/oik.07645

Sarremejane, R., Truchy, A., McKie, B.G., Mykrä, H., Johnson, R.K., Huusko, A., Sponseller, R.A., and Muotka, T., 2021a. Stochastic processes and ecological connectivity drive stream invertebrate community responses to short-term drought. Journal of Animal Ecology 90: 886–898. https://doi.org/10.1111/1365-2656.13417

Sarremejane, R., Stubbington, R., England, J., Sefton, C.E., Eastman, M., Parry, S. and Ruhi, A., 2021b. Drought effects on invertebrate metapopulation dynamics and quasi-extinction risk in an intermittent river network. Global Change Biology 27: 4024–4039. https://doi.org/10.1111/gcb.15720

Sarremejane, R., Messager, M.L. and Datry, T., 2022. Drought in intermittent river and ephemeral stream networks. Ecohydrology 15: e2390. https://doi.org/10.1002/eco.2390

Sauquet, E., Beaufort, A., Sarremejane, R. and Thirel, G., 2021. Predicting flow intermittence in France under climate change. Hydrological Sciences Journal 66: 2046–2059. https://doi.org/10.1080/02626667.2021.1963444

Schaum, C.E., Student Research Team, ffrench-Constant, R., Lowe, C., Ólafsson, J.S., Padfield, D. and Yvon-Durocher, G., 2018. Temperature-driven selection on metabolic traits increases the strength of an algal–grazer interaction in naturally warmed streams. Global Change Biology 24: 1793–1803. https://doi.org/10.1111/gcb.14033

Scheffer, M., Carpenter, S., Foley, J.A., Folke, C. and Walker, B., 2001. Catastrophic shifts in ecosystems. Nature 413: 591–596. https://doi.org/10.1038/35098000

Sefton, C.E., Parry, S., England, J. and Angell, G., 2019. Visualising and quantifying the variability of hydrological state in intermittent rivers. Fundamental and Applied Limnology, 193: 21–38. https://doi.org/10.1127/fal/2019/1149

Seneviratne, S.I. et al. 2012: Changes in climate extremes and their impacts on the natural physical environment, pp. 109–230 in: Field, C.B. et al. (Eds.). Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation: A Special Report of Working Groups I and II of the Intergovernmental Panel on Climate Change (IPCC). Cambridge University Press. Available at: https://www.ipcc.ch/site/assets/uploads/2018/03/SREX-Chap3_FINAL-1.pdf

Shaffrey, L., 2023. Atmospheric circulation, dry spells, and drought in the UK. Environment Agency report series: Review of the state of research on drought.

Shand, P., Edmunds, W.M., Lawrence, A.R., Smedley, P.L. and Burke, S., 2007. The natural (baseline) quality of groundwater in England and Wales. British Geological Survey Research Report No. RR/07/06. Environment Agency/British Geological Survey. Available at: https://core.ac.uk/download/pdf/62722.pdf

Shaw, G., 1979. Prey selection by breeding dippers. Bird Study 26: 66–67. https://doi.org/10.1080/00063657909476619

Slette, I.J., Post, A.K., Awad, M., Even, T., Punzalan, A., Williams, S., Smith, M.D. and Knapp, A.K., 2019. How ecologists define drought, and why we should do better. Global Change Biology 25: 3193–3200. https://doi.org/10.1111/gcb.14747

Smith, M.D., 2011. An ecological perspective on extreme climatic events: a synthetic definition and framework to guide future research. Journal of Ecology 99: 656–663. https://doi.org/10.1111/j.1365-2745.2011.01798.x

Solomon, D.J. and Paterson, D., 1980. Influence of natural and regulated streamflow on survival of brown trout (Salmo trutta L.) in a chalkstream. Environmental Biology of Fishes 5: 379–382. https://doi.org/10.1007/BF00005191

Solomon, D.J. and Sambrook, H.T., 2004. Effects of hot dry summers on the loss of Atlantic salmon, Salmo salar, from estuaries in South West England. Fisheries Management and Ecology 11: 353–363. https://doi.org/10.1111/j.1365-2400.2004.00403.x

Solomon, D.J. and Templeton, R.G., 1976. Movements of brown trout Salmo trutta L. in a chalk stream. Journal of Fish Biology 9: 411–423. https://doi.org/10.1111/j.1095-8649.1976.tb04690.x

Sousa, R., Ferreira, A., Carvalho, F., Lopes-Lima, M., Varandas, S. and Teixeira, A., 2018. Die-offs of the endangered pearl mussel Margaritifera margaritifera during an extreme drought. Aquatic Conservation: Marine and Freshwater Ecosystems 28: 1244–1248. https://doi.org/10.1002/aqc.2945

Spinoni, J., Naumann, G., Vogt, J. and Barbosa, P., 2015. European drought climatologies and trends based on a multi-indicator approach. Global and Planetary Change 127 50–57. https://doi.org/10.1016/j.gloplacha.2015.01.012

Spinoni, J., Vogt, J.V., Naumann, G., Barbosa, P. and Dosio, A., 2018. Will drought events become more frequent and severe in Europe? International Journal of Climatology 38: 1718–1736. https://doi.org/10.1002/joc.5291

Steward, A.L., Datry, T. and Langhans, S.D., 2022. The terrestrial and semi-aquatic invertebrates of intermittent rivers and ephemeral streams. Biological Reviews 97: 1408–1425. https://doi.org/10.1111/brv.12848

Stoelzle, M., Stahl, K. Morhard, A. and Weiler M., 2014. Streamflow sensitivity to drought scenarios in catchments with different geology. Geophysical Research Letters 41: 6174–6183. https://doi.org/10.1002/2014GL061344

Storey, R.G. and Quinn, J.M., 2011. Life histories and life history strategies of invertebrates inhabiting intermittent streams in Hawke's Bay, New Zealand. New Zealand Journal of Marine and Freshwater Research 45: 213–230. https://doi.org/10.1080/00288330.2011.554988

Strachan, S.R., Chester, E.T. and Robson, B.J., 2015. Freshwater invertebrate life history strategies for surviving desiccation. Springer Science Reviews 3: 57–75. https://doi.org/10.1007/s40362-015-0031-9

Strachan, S.R., Chester, E.T. and Robson, B.J., 2016. Habitat alters the effect of false starts on seasonal-wetland invertebrates. Freshwater Biology 61: 680–692. https://doi.org/10.1111/fwb.12738

Stradmeyer, L. and Thorpe, J.E., 1987. Feeding behaviour of wild Atlantic salmon, Salmo salar L., parr in mid-to late summer in a Scottish river. Aquaculture Research 18: 33-49. https://doi.org/10.1111/j.1365-2109.1987.tb00123.x

Stradmeyer, L., Höjesjö, J., Griffiths, S. W., Gilvear, D. J., & Armstrong, J. D. (2008). Competition between brown trout and Atlantic salmon parr over pool refuges during rapid dewatering. Journal of Fish Biology 72: 848–860. https://doi.org/10.1111/j.1095-8649.2007.01767.x

Stubbington, R., 2012. The hyporheic zone as an invertebrate refuge: a review of variability in space, time, taxa and behaviour. Marine and Freshwater Research 63: 293–311. https://doi.org/10.1071/MF11196

Stubbington, R. and Datry, T., 2013. The macroinvertebrate seedbank promotes community persistence in temporary rivers across climate zones. Freshwater Biology 58: 1202–1220. https://doi.org/10.1111/fwb.12121

Stubbington, R. and Wood, P.J., 2013. Benthic and interstitial habitats of a lentic spring as invertebrate refuges during supra-seasonal drought. Fundamental and Applied Limnology, 182: 61–73. http://dx.doi.org/10.1127/1863-9135/2013/0409

Stubbington, R., Wood, P.J. and Boulton, A.J., 2009a. Low flow controls on benthic and hyporheic macroinvertebrate assemblages during supra-seasonal drought. Hydrological Processes 23: 2252–2263. https://doi.org/10.1002/hyp.7290

Stubbington, R., Greenwood, A.M., Wood, P.J., Armitage, P.D., Gunn, J. and Robertson, A.L., 2009b. The response of perennial and temporary headwater stream invertebrate communities to hydrological extremes. Hydrobiologia 630: 299–312. https://doi.org/10.1007/s10750-009-9823-8

Stubbington, R., Wood, P.J. and Reid, I., 2011. Spatial variability in the hyporheic zone refugium of temporary streams. Aquatic Sciences 73: 499–511. https://doi.org/10.1007/s00027-011-0203-x

Stubbington, R., Boulton, A.J., Little, S. and Wood, P.J., 2015. Changes in invertebrate assemblage composition in benthic and hyporheic zones during a severe supraseasonal drought. Freshwater Science 34: 344–354. https://doi.org/10.1086/679467

Stubbington, R., Gunn, J., Little, S., Worrall, T.P. and Wood, P.J., 2016. Macroinvertebrate seedbank composition in relation to antecedent duration of drying and multiple wet-dry cycles in a temporary stream. Freshwater Biology 61: 1293–1307. https://doi.org/10.1111/fwb.12770

Stubbington, R., Bogan, M.T., Bonada, N., Boulton, A.J., Datry, T., Leigh, C. and Vander Vorste, R., 2017. The biota of intermittent rivers and ephemeral streams: aquatic invertebrates, pp. 217–243 in: Datry, T., Bonada, N. and Boulton, A.J. (Eds.). Intermittent Rivers and Ephemeral Streams. Academic Press. https://doi.org/10.1016/B978-0-12-803835-2.00007-3

Stubbington R, Dimon J, England J and Watts G. 2022. Chalk streams of the future: the effects of climate change on biodiversity in England's iconic river ecosystems. https://doi.org/10.6084/m9.figshare.21663269

Symons, P.E.K. and Héland, M., 1978. Stream habitats and behavioral interactions of underyearling and yearling Atlantic salmon (Salmo salar). Journal of the Fisheries Board of Canada 35: 175–183. https://doi.org/10.1139/f78-029

Tapia, G., Bass, J.A.B. and House, A., 2018. Further occurrence records for the Winterbourne Stonefly Nemoura lacustris Pictet, 1865 (Plecoptera: Nemouridae). Entomologist's Monthly Magazine 154: 60–64. https://doi.org/10.31184/M00138908.1541.3915

Taverny, C., Lassalle, G., Ortusi, I., Roqueplo, C., Lepage, M. and Lambert, P., 2012. From shallow to deep waters: habitats used by larval lampreys (genus Petromyzon and Lampetra) over a western European basin. Ecology of Freshwater Fish 21: 87–99. https://doi.org/10.1111/j.1600-0633.2011.00526.x

Tesch, F.W., 2003. The Eel. 5th ed. 416 pp. Oxford, Blackwell Publishing Company.

Thoms, M.C., 2006. Variability in riverine ecosystems. River Research and Applications 22: 115–121. https://doi.org/10.1002/rra.900

Tijdeman, E., Hannaford, J. and Stahl, K., 2018. Human influences on streamflow drought characteristics in England and Wales. Hydrology and Earth System Sciences 22: 1051–1064. https://doi.org/10.5194/hess-22-1051-2018

Timoner, X., Acuña, V., Frampton, L., Pollard, P., Sabater, S. and Bunn, S.E., 2014. Biofilm functional responses to the rehydration of a dry intermittent stream. Hydrobiologia 727: 185–195. https://doi.org/10.1007/s10750-013-1802-4

Tonkin, J.D., Poff, N.L., Bond, N.R., Horne, A., Merritt, D., Reynolds, L.V., Olden, J.D., Ruhi, A. and Lytle, D.A., 2019. Prepare river ecosystems for an uncertain future. Nature 570: 301–303. https://doi.org/10.1038/d41586-019-01877-1

Tornwall, B.M., Swan, C.M. and Brown, B.L., 2017. Manipulation of local environment produces different diversity outcomes depending on location within a river network. Oecologia 184: 663–674. https://doi.org/10.1007/s00442-017-3891-7

Tramblay, Y., Rutkowska, A., Sauquet, E., Sefton, C., Laaha, G., Osuch, M., Albuquerque, T., Alves, M.H., Banasik, K., Beaufort, A. and Brocca, L., 2021. Trends in flow intermittence for European rivers. Hydrological Sciences Journal 66: 37–49. https://doi.org/10.1080/02626667.2020.1849708

Truchy, A., Sarremejane, R., Muotka, T., Mykrä, H., Angeler, D.G., Lehosmaa, K., Huusko, A., Johnson, R.K., Sponseller, R.A. and McKie, B.G., 2020. Habitat patchiness, ecological connectivity and the uneven recovery of boreal stream ecosystems from an experimental drought. Global Change Biology 26: 3455–3472. https://doi.org/10.1111/gcb.15063

Turley, M.D., Bilotta, G.S., Krueger, T., Brazier, R.E. and Extence, C.A., 2015. Developing an improved biomonitoring tool for fine sediment: combining expert knowledge and empirical data. Ecological Indicators 54: 82–86. https://doi.org/10.1016/j.ecolind.2015.02.011

Turley, M.D., Bilotta, G.S., Chadd, R.P., Extence, C.A., Brazier, R.E., Burnside, N.G. and Pickwell AG. 2016. A sediment-specific family-level biomonitoring tool to identify the impacts of fine sediment in temperate rivers and streams. Ecological Indicators 70: 151–165. https://doi.org/10.1016/j.ecolind.2016.05.040

Turner, S., Barker, L.J., Hannaford, J., Muchan, K., Parry, S. and Sefton, C., 2021. The 2018/2019 drought in the UK: a hydrological appraisal. Weather 76: 248–253. https://doi.org/10.1002/wea.4003

Vadher, A.N., Stubbington, R. and Wood, P.J., 2015. Fine sediment reduces vertical migrations of Gammarus pulex (Crustacea: Amphipoda) in response to surface water loss. Hydrobiologia 753: 61–71. https://doi.org/10.1007/s10750-015-2193-5

Van der Bolt, B., van Nes, E.H. and Scheffer, M., 2021. No warning for slow transitions. Journal of the Royal Society Interface 18: 20200935. https://doi.org/10.1098/rsif.2020.0935

Van Loon, A.F., 2015. Hydrological drought explained. Wiley Interdisciplinary Reviews: Water 2: 359–392. https://doi.org/10.1002/wat2.1085

Van Loon, A.F., Gleeson, T., Clark, J., Van Dijk, A.I., Stahl, K., Hannaford, J., Di Baldassarre, G., Teuling, A.J., Tallaksen, L.M., Uijlenhoet, R. and Hannah, D.M., 2016. Drought in the Anthropocene. Nature Geoscience 9: 89–91. https://doi.org/10.1038/ngeo2646

Van Loon, A.F., et al., 2022. Streamflow droughts aggravated by human activities despite management. Environmental Research Letters 17: 044059. https://doi.org/10.1088/1748-9326/ac5def

van Nes, E.H., Arani, B.M., Staal, A., van der Bolt, B., Flores, B.M., Bathiany, S. and Scheffer, M., 2016. What do you mean, 'tipping point'? Trends in Ecology & Evolution 31: 902–904. https://doi.org/10.1016/j.tree.2016.09.011

Vander Vorste, R., Malard, F. and Datry, T., 2016a. Is drift the primary process promoting the resilience of river invertebrate communities? A manipulative field experiment in an intermittent alluvial river. Freshwater Biology 61: 1276–1292. https://doi.org/10.1111/fwb.12658

Vander Vorste, R., Mermillod-Blondin, F., Hervant, F., Mons, R., Forcellini, M. and Datry, T., 2016b. Increased depth to the water table during river drying decreases the resilience of Gammarus pulex and alters ecosystem function. Ecohydrology 9: 1177–1186. https://doi.org/10.1002/eco.1716

Vander Vorste, R., Obedzinski, M., Nossaman Pierce, S., Carlson, S.M. and Grantham, T.E., 2020. Refuges and ecological traps: Extreme drought threatens persistence of an endangered fish in intermittent streams. Global Change Biology 26: 3834–3845. https://doi.org/10.1111/gcb.15116

Verdonschot, R.C.M., van Oosten-Siedlecka, A.M., ter Braak, C.J.F. and Verdonschot, P.F.M., 2015. Macroinvertebrate survival during cessation of flow and streambed drying in a lowland stream. Freshwater Biology 60: 282–296. https://doi.org/10.1111/fwb.12479

Vehanen, T., Huusko, A., Mäki-Petäys, A., Louhi, P., Mykrä, H. and Muotka, T., 2010. Effects of habitat rehabilitation on brown trout (Salmo trutta) in boreal forest streams. Freshwater Biology 55: 2200–2214. https://doi.org/10.1111/j.1365-2427.2010.02467.x

Wada, Y., Van Beek, L.P., Wanders, N. and Bierkens, M.F., 2013. Human water consumption intensifies hydrological drought worldwide. Environmental Research Letters 8: 034036. http://dx.doi.org/10.1088/1748-9326/8/3/034036

Wade, A.J., Whitehead, P.G., Hornberger, G.M. and Snook, D.L., 2002. On modelling the flow controls on macrophyte and epiphyte dynamics in a lowland permeable catchment: the River Kennet, southern England. Science of the Total Environment 282: 375–393. https://doi.org/10.1016/S0048-9697(01)00925-1

Walker, D.W. and Van Loon, A.F., 2023. Droughts are coming on faster. Science 380: 130–132. https://doi.org/10.1126/science.adh3097

Watts, G., Battarbee, R.W., Bloomfield, J.P., Crossman, J., Daccache, A., Durance, I., Elliott, J.A., Garner, G., Hannaford, J., Hannah, D.M. and Hess, T., 2015. Climate change and water in the UK–past changes and future prospects. Progress in Physical Geography 39: 6–28. https://doi.org/10.1177/0309133314542957

Weatherley, N.S., Campbell-Lendrum, E.W. and Ormerod, S.J., 1991. The growth of brown trout (Salmo trutta) in mild winters and summer droughts in upland Wales: model validation and preliminary predictions. Freshwater Biology 26: 121–131. https://doi.org/10.1111/j.1365-2427.1991.tb00514.x

Webb, R.H. and Leake, S.A., 2006. Ground-water surface-water interactions and long-term change in riverine riparian vegetation in the southwestern United States. Journal of Hydrology 320: 302–323. https://doi.org/10.1016/j.jhydrol.2005.07.022

Welton, J.S., Bass, J.A.B., Ladle, M. and Merrett, W.J., 1987. Distribution of oviposition sites and characteristics of egg development in the ‘Blandford fly' Simulium posticatum (Diptera: Simuliidae). Journal of Applied Ecology 24: 865–879. https://doi.org/10.2307/2403986

Wessex Water, 2022. Drought Plan: Final Plan: Wessex Water: June 2022. Available via: https://www.wessexwater.co.uk/environment/water-resources/drought-plan

Westwood, C.G., Teeuw, R.M., Wade, P.M., Holmes, N.T.H. and Guyard, P., 2006a. Influences of environmental conditions on macrophyte communities in drought-affected headwater streams. River Research and Applications 22: 703–726. https://doi.org/10.1002/rra.934

Westwood, C.G., Teeuw, R.M., Wade, P.M. and Holmes, N.T.H., 2006b. Prediction of macrophyte communities in drought-affected groundwater-fed headwater streams. Hydrological Processes 20: 127–145. https://doi.org/10.1002/hyp.5907

Westwood, C.G., England, J., Dunbar, M.J., Holmes, N.T., Leeming, D.J. and Hammond, D., 2017. An approach to setting ecological flow thresholds for southern English chalk streams. Water and Environment Journal 31: 528–536. https://doi.org/10.1111/wej.12275

Westwood, C.G., England, J., Johns, T. and Stubbington, R., 2020. A revised classification of temperate lowland groundwater-fed headwater streams based on their flora. Water and Environment Journal 34: 573–585. https://doi.org/10.1111/wej.12561

Westwood, C.G., England, J., Hayes, C., Johns, T. and Stubbington, R., 2021. The Plant Flow Index: A new method to assess the hydroecological condition of temporary rivers and streams. Ecological Indicators 120: 106964. https://doi.org/10.1016/j.ecolind.2020.106964

White, J.C., House, A., Punchard, N., Hannah, D.M., Wilding, N.A. and Wood, P.J., 2018. Macroinvertebrate community responses to hydrological controls and groundwater abstraction effects across intermittent and perennial headwater streams. Science of the Total Environment 610: 1514–1526. https://doi.org/10.1016/j.scitotenv.2017.06.081

Wilby, R.L. and Johnson, M.F., 2020. Climate variability and implications for keeping rivers cool in England. Climate Risk Management 30: 100259. https://doi.org/10.1016/j.crm.2020.100259

Wilby, R.L., Cranston, L.E. and Darby, E.J., 1998. Factors governing macrophyte status in Hampshire chalk streams: implications for catchment management. Water and Environment Journal 12: 179–187. https://doi.org/10.1111/j.1747-6593.1998.tb00170.x

Williams, G.M.D., 2016. Impact of drought on stream ecosystem structure and functioning. PhD thesis. University of Birmingham. Available at: https://etheses.bham.ac.uk/id/eprint/6950/1/Williams16PhD.pdf

Williams, A.P., Cook, E.R., Smerdon, J.E., Cook, B.I., Abatzoglou, J.T., Bolles, K., Baek, S.H., Badger, A.M. and Livneh, B., 2020. Large contribution from anthropogenic warming to an emerging North American megadrought. Science 368: 314–318. https://doi.org/10.1126/science.aaz9600

Wohl, E., Lininger, K.B. and Scott, D.N., 2018 River beads as a conceptual framework for building carbon storage and resilience to extreme climate events into river management. Biogeochemistry 141: 365–383. https://doi.org/10.1007/s10533-017-0397-7

Wood, P.J. and Armitage, P.D., 1997. Biological effects of fine sediment in the lotic environment. Environmental Management 21: 203–217. https://doi.org/10.1007/s002679900019

Wood, P.J. and Petts, G.E., 1994. Low flows and recovery of macroinvertebrates in a small regulated chalk stream. Regulated Rivers: Research & Management 9: 303–316. https://doi.org/10.1002/rrr.3450090410

Wood, P.J. and Petts, G.E., 1999. The influence of drought on chalk stream macroinvertebrates. Hydrological Processes 13: 387–399. https://doi.org/10.1002/(SICI)1099-1085(19990228)13:3%3C387::AID-HYP745%3E3.0.CO;2-R

Wood, P.J., Agnew, M.D. and Petts, G.E., 2000. Flow variations and macroinvertebrate community responses in a small groundwater-dominated stream in south-east England.

Hydrological Processes 14: 3133–3147. https://doi.org/10.1002/1099-1085(200011/12)14:16/17%3C3133::AID-HYP138%3E3.0.CO;2-J

Wood, P.J., Hannah, D.M., Agnew, M.D. and Petts, G.E., 2001. Scales of hydroecological variability within a groundwater-dominated stream. Regulated Rivers: Research & Management 17: 347–367. https://doi.org/10.1002/rrr.658

Wood, P.J., Boulton, A.J., Little, S. and Stubbington, R., 2010. Is the hyporheic zone a refugium for macroinvertebrates during severe low flow conditions? Fundamental and Applied Limnology 176: 377–390. https://doi.org/10.1127/1863-9135/2010/0176-0377

Woodhouse, C.A. and Overpeck, J.T., 1998. 2000 years of drought variability in the central United States. Bulletin of the American Meteorological Society 79: 2693–2714. https://doi.org/10.1175/1520-0477(1998)079%3C2693:YODVIT%3E2.0.CO;2

Woodward, G., Papantoniou, G., Edwards, F. and Lauridsen, R.B., 2008. Trophic trickles and cascades in a complex food web: impacts of a keystone predator on stream community structure and ecosystem processes. Oikos 117: 683–692. https://doi.org/10.1111/j.0030-1299.2008.16500.x

Woodward, G., Brown, L.E., Edwards, F.K., Hudson, L.N., Milner, A.M., Reuman, D.C. and Ledger, M.E., 2012. Climate change impacts in multispecies systems: drought alters food web size structure in a field experiment. Philosophical Transactions of the Royal Society B: Biological Sciences 367: 2990–2997. https://doi.org/10.1098/rstb.2012.0245

Woodward, G., Morris, O., Barquín, J., Belgrano, A., Bull, C., de Eyto, E., Friberg, N., Guðbergsson, G., Layer-Dobra, K., Lauridsen, R.B. and Lewis, H.M., 2021. Using food webs and metabolic theory to monitor, model, and manage Atlantic Salmon—A keystone species under threat. Frontiers in Ecology and Evolution 9: 675261. https://doi.org/10.3389/fevo.2021.675261

Wright, J.F., 1992. Spatial and temporal occurrence of invertebrates in a chalk stream, Berkshire, England. Hydrobiologia 248: 11–30. https://doi.org/10.1007/BF00008882

Wright, J.F., 1999. An investigation of ecological change in the rivers Kennet and Lambourn. Environment Agency. Available at: https://nora.nerc.ac.uk/id/eprint/507655/1/N507655CR.pdf

Wright, J.F. and Berrie, A.D., 1987. Ecological effects of groundwater pumping and a natural drought on the upper reaches of a chalk stream. Regulated Rivers: Research & Management 1: 145–160. https://doi.org/10.1002/rrr.3450010205

Wright, J.F. and Symes, K.L., 1999. A nine-year study of the macroinvertebrate fauna of a chalk stream. Hydrological Processes 13: 371–385. https://doi.org/10.1002/(SICI)1099-1085(19990228)13:3%3C371::AID-HYP744%3E3.0.CO;2-C

Wright, J.F., Hiley, P.D. and Berrie, A.D., 1981. A nine-year study of the life cycle of Ephemera danica Mull. (Ephemeridae. Ephemeroptera) in the River Lambourn. Ecological Entomology 6: 321–331. https://doi.org/10.1111/j.1365-2311.1981.tb00619.x

Wright, J.F., Blackburn, J.H., Gunn, R.J.M., Symes, K.L. and Bowker, J., 1998. A scoping study on the Ephemeroptera of southern chalk streams. Environment Agency. Available at: https://nora.nerc.ac.uk/id/eprint/502099/1/N502099RE.pdf

Wright, J.F., Gunn, R.J.M., Winder, J.M., Blackburn, J.H. and Wiggers, R., 2000. The response of chalk stream macroinvertebrates to a prolonged drought: the value of a long-term dataset. Internationale Vereinigung für Theoretische und Angewandte Limnologie: Verhandlungen 27: 912–915. https://doi.org/10.1080/03680770.1998.11901371

Wright, J.F., Gunn, R.J.M., Winder, J.M., Wiggers, R., Vowles, K., Clarke, R.T. and Harris, I., 2002. A comparison of the macrophyte cover and macroinvertebrate fauna at three sites on the River Kennet in the mid 1970s and late 1990s. Science of the Total Environment 282: 121–142. https://doi.org/10.1016/S0048-9697(01)00948-2

Wright, J.F., Clarke, R.T., Gunn, R.J., Winder, J.M., Kneebone, N.T. and Davy-Bowker, J., 2003. Response of the flora and macroinvertebrate fauna of a chalk stream site to changes in management. Freshwater Biology 48: 894–911. https://doi.org/10.1046/j.1365-2427.2003.01058.x

Wright, J.F., Clarke, R.T., Gunn, R.J.M., Kneebone, N.T. and Davy-Bowker, J., 2004. Impact of major changes in flow regime on the macroinvertebrate assemblages of four chalk stream sites, 1997–2001. River Research and Applications 20: 775–794. https://doi.org/10.1002/rra.790

Young, B.A., Norris, R.H. and Sheldon, F., 2011. Is the hyporheic zone a refuge for macroinvertebrates in drying perennial streams? Marine and Freshwater Research 62: 1373–1382. https://doi.org/10.1071/MF11060

Yuan, X., Wang, Y., Ji, P., Wu, P., Sheffield, J. and Otkin, J.A., 2023. A global transition to flash droughts under climate change. Science 380: 187–191. https://doi.org/10.1126/science.abn6301

Zipper, S., Popescu, I., Compare, K., Zhang, C. and Seybold, E.C., 2022. Alternative stable states and hydrological regime shifts in a large intermittent river. Environmental Research Letters 17: 074005. https://doi.org/10.1088/1748-9326/ac7539

Appendices

Appendix 1. Research characterizing the effects of drought in England's chalk streams

A considerable body of research has documented responses to drought in the chalk rivers and streams of southern England, although interpretation of all resultant research findings can be confounded by concurrent alteration of river flow by groundwater and surface abstraction and/or low-flow alleviation schemes involving flow augmentation (e.g. Solomon and Paterson 1980; Wright and Berrie 1987; Holmes 1999; Perrow et al. 2007).

Five notable sets of studies are among those characterizing aquatic invertebrate community responses to and recovery after drought:

1. Field-based studies documented responses to the 1973, 1976 and/or 1997 droughts on the perennial River Lambourn and its Winterbourne Stream, Berkshire (Wright and Berrie 1987; Wright 1992; Wright and Symes 1999; Wright et al. 2000, 2002, 2004) as well as the near-perennial Waterston Stream, a tributary of the River Piddle, Dorset (Ladle and Bass 1981).

2. Long-term, river-scale field research characterized perennial and near-perennial reaches on the Little Stour, Kent (Wood and Petts 1994, 1999; Wood et al. 2000; Stubbington et al. 2009a; Wood et al. 2010; Stubbington and Wood 2013; Stubbington et al. 2015).

3. Long-term, regional-scale, field-based studies have examined perennial, near-perennial and seasonally intermittent (i.e. winterbourne) reaches across multiple streams in the Colne and Lee catchments, Hertfordshire (Sarremejane et al. 2019, 2020, 2021b), supported by exceptional mapping of wet and dry in-channel habitat conditions (Sefton et al. 2019).

4. Manipulative experimental studies determined how short (six-day), high (monthly) and low (quarterly) frequency dry phases affected food web structure in channels next to perennial reaches of the Mill Stream, a side branch of the River Frome, Dorset (Ledger et al. 2011, 2012, 2013; Woodward et al. 2012; Lu et al. 2016; Leigh et al. 2019).

5. Manipulative experimental studies characterized taxonomic and functional community responses to year-long high, moderate and low magnitude drought conditions (Aspin et al. 2018, 2019a,b) in channels adjacent to perennial reaches of the Candover Brook, a tributary of the River Itchen, Hampshire, with Williams (2016) using the same channels to investigate interactions among drought, temperature and sedimentation.

This research has focused mainly on invertebrate communities in surface (i.e. benthic) sediments, with the 2009–15 Little Stour studies also documenting benthic and groundwater invertebrates in the subsurface sediments of the hyporheic zone. In addition, although not the focus of this essay, other biotic groups have been characterized in chalk streams, supporting understanding of whole-ecosystem responses:

The effects of drought on chalk stream **macrophytes** (aquatic plants) and terrestrial vegetation have also received notable attention, including:

- Wright and colleagues' research described in point 1 above entailed characterization of invertebrate communities on habitats including Ranunculus (water crowfoot), Berula (water parsnip) and Callitriche (water starwort). These studies also note (but do not characterize) biofilms of epiphytic algae, and provide insight into the links between biofilm, plant and invertebrate responses to drought.
- A 10-year study recorded aquatic and terrestrial plant community responses to flow variability, including that experienced during the 1989–92 groundwater drought and subsequent 1996–98 drought, across an extensive range of chalk headwater streams in the Thames catchment (Holmes 1999; Westwood et al. 2006a,b). Westwood et al. (2017, 2020, 2021) then continued this work, creating a classification and indicators to document community responses to flow variability and drought.
- Wilby et al. (1998) report macrophyte survey data collected from sites along Hampshire's Rivers Test and Itchen, from the headwaters to the lower reaches, on 17 dates in 1991–1996, encompassing before, during and after the dry, hot summers of 1994 and 1995.
- Ham et al. (1981) mapped sediment and macrophyte composition on an unshaded section of the River Lambourn, Berkshire in 1971–76, including the 1976 drought.

Relatively few studies have documented the effects of drought on fish in chalk streams, with a greater body of research instead reporting general relationships with flow and temperature from which species-specific and community-level drought effects may be inferred (e.g. for Atlantic Salmon Salmo salar, Mann 1989; brown trout Salmo trutta, Mann et al. 1989; eel Anguilla anguilla, Riley et al. 2011; grayling Thymallus thymallus, Bašić et al. 2018, Marsh et al. 2021; and communities, Prenda et al. 1997, Nunn et al. 2010). The few studies directly considering drought include Wright and Berrie (1987), who report the effects of drought and a flow augmentation scheme in the River Lambourn, Berkshire; and Riley et al. (2009), who experimentally created drought-like conditions in the Brandy stream, a 400-m tributary of the River Itchen, Hampshire.

Experimental research has also document responses of the assemblage of microorganisms (including diatoms) in biofilms to drought and related variables in chalk stream: studies in the stream mesocosm fed by the Mill Stream (described above) considered biofilms as well as invertebrates (Ledger et al. 2008); and Jarvie and colleagues (e.g. Flynn et al. 2002; Jarvie et al. 2002) characterized relationships with flow and solar radiation for experimental biofilm communities in the River Kennet, Wiltshire. Scattered field-based observations of such assemblages include Moore's (1977) study of sediment-associated and epiphytic algal communities in relation to flow, temperature and light (but not drought) in the River Wylye, Wiltshire; and qualitative observations of epiphytic algal growth in studies focused on macrophyte assemblages (Wright and Berrie 1987; Wright and Symes 1999). Characterization of phytoplankton is rare and drought-specific studies are lacking, but characterization of assemblages in the River Thames include documentation of relationships with flow and temperature based on field observations (Bowes et al. 2016) and associated predictive modelling (Bussi et al. 2016).

Appendix 2. Hydrological drought descriptors

Hydrological drought is a deficit in streamflow. It can be further characterized by metrics that describe its frequency, magnitude, duration, severity, timing, and the rate at which conditions change.

- Drought increases the **frequency** with which streamflow (or sediment moisture levels) falls below a particular threshold, such as the Q95 (the discharge equalled or exceeded 95% of the time). (Note that droughts can also be described by their frequency, or the return period [e.g. 1 in 5, 10 or 20 years] for hydrological conditions of a particular duration, magnitude or spatial extent [Vicente-Serrano et al. 2019] although climate change is causing ongoing changes in such metrics and their reliability.)
- A drought can be described by its **magnitude** i.e. the extent to which streamflow, groundwater or moisture levels drop below the long-term average (or other defining threshold). Sarremejane et al. (2022) distinguish between drought magnitude (or intensity) and severity as the average and cumulative deficit in streamflow, respectively. As such, a drought's severity is determined by both its average magnitude and its duration (Keyantash and Dracup 2004).
- A drought can be described by its **duration**, and increases the duration for which streamflow (or sediment moisture levels) remain below a particular threshold, such as the Q95. Drought durations may be restricted to single seasons, span multiple seasons (i.e. supra-seasonal drought) or even last for multiple years.
- A drought can be described by its timing, including single-season summer droughts and winter droughts.
- Drought onset and termination, as well as shifts between local-scale hydrological conditions, can be described by their **rate of change**, with rapid shifts associated with greater ecological impacts.

Table 1 in Sarremejane et al. (2022) provides further details of these metrics.

Appendix 3. Species of conservation concern within drought-intolerant families

The DEHLI (Drought Effect of Habitat Loss on Invertebrates) index was developed to characterize macroinvertebrate community responses to drought-driven changes in habitat availability at the family level, despite the contrasting habitat requirements of species and genera within many families (Chadd et al. 2017). The families below are those with the highest (9–10) "drought intolerance score (DIS)" in Chadd et al. (2017), where the highest DIS indicate association with habitats typically lost first as the typical sequence of drought-driven changes to habitat conditions unfolds (see section 4), i.e. rheophilic (fast-flow-loving) 'EPT' (Ephemeroptera [mayfly], Plecoptera [stonefly] and Trichoptera [caddisfly]) families. As such, an increasingly wide range of other families and species may be at comparable risk if drought causes loss of lateral or longitudinal connectivity to decline, including formation, then isolation, then drying of pools (Chadd et al. 2017).

The species of conservation concern within each drought-sensitive family are either listed as threatened in the most recent Great Britain Species Status Review (Macadam 2015, 2016; Wallace 2016) and—in one case—is endemic to Great Britain (Craig Macadam, Conservation Director, Buglife; pers. comm.).

Family	Species	Status[1]	Notes
Ameletidae	Ameletus inopinatus	LC; NS	Headwater species
Chloroperlidae	Xanthoperla apicalis	EX	
Heptageniidae	Electrogena affinis	DD; NR	
	Heptagenia longicauda	EX	
	Kageronia fuscogrisea	LC; NS	
	Rhithrogena germanica	LC; NS	
Taeniopterygidae	Brachyptera putata	LC; NS	Endemic
	Rhabdiopteryx acuminata	VU; NR	

[1]DD = Data Deficient; EX = Extinct; NR = Nationally Rare; NS = Nationally Scarce; LC = Least Concern; VU = Vulnerable

H: Soils and Drought: An Essay

A.M. Tye[1], D.A. Robinson[2], S. Reinsch [2], A. Keith[2] and Parry, S. [2]

[1]British Geological Survey (BGS)

[2]UK Centre for Ecology & Hydrology (UKCEH)

Overview

- The summer drought across the UK in 2022 produced significant speculation concerning how its termination may impact and interact with the soil resource. Two key areas of concern were those related to the potential of intense convective precipitation, falling on sun-baked soil and leading to (i) excessive runoff and flooding and (ii) erosion of the soil resource resulting from this excess runoff. The scientific literature was reviewed to develop a coherent understanding of the wider range of impacts of drought on soils for the UK. In particular, the review focuses on 'meteorological drought' as this is when soils experience long dry periods followed by re-wetting. Key themes drawn out from the review are:
- There is generally very little information pertaining specifically to the responses of UK soils to drought. This is because unlike surface and groundwater, few long-term soil monitoring programs are in place, measuring properties relevant to drought. In addition, drought in the UK is unpredictable making it harder to plan specific experiments to examine its impacts. Information has been gathered from international studies, climate manipulation studies and particularly from the more often studied role of wetting and drying cycles on soil properties.
- The key process for soils when rewetting after drought is whether infiltration > precipitation. When infiltration > precipitation the soils should be able to absorb the precipitation. When precipitation > infiltration, potential runoff and erosion may occur, leading to loss of the soil resource and potential degradation of surface water quality.
- The literature review demonstrates the important interactions between soil biology, chemistry, and physics with respect to drought. In many smectitic clay soils highly visible cracking occurs as drying takes place. However, the role of soil biology on soil physical structure, particularly aggregate stability, is one of the most important impacts of drought on soils. The decrease of biological activity which helps maintain soil structure, is a key impact leading to potential slaking, soil crust formation, and potentially decreasing infiltration upon rewetting. Changes in the structure of soils may lead to changes in biogeochemical cycling of C, N and P through the exposure of aggregate protected organic matter. Other biogeochemical cycles may also be impacted, particularly those which are redox sensitive and where droughts produce the lowering of soil moisture and water tables.
- When droughts end, results from studies suggest that soil moisture will increase rapidly, eventually becoming recharged as more normal precipitation

patterns return. However, the understanding of how the shallow water table responds with time is limited. The shallow water table is a potential source of water to the soil column if drought returns after breaking and more information would be beneficial.

- Key gaps in our understanding are related to the resilience, response, and recovery of soil systems. Information is required regarding soil biological communities, soil water repellency and the extent that shallow ground water may buffer the soil moisture system. In addition, further gaps in our knowledge exist regarding multiple stressors (heat, moisture) and the impacts of successive extreme events on soil systems (e.g. drought, flood).

Summary

The summer drought across the UK in 2022 produced significant speculation concerning how its termination may impact and interact with the soil resource. Two key areas of concern raised in the media were those related to the potential of intense convective precipitation, falling on sun-baked soil and inducing (i) excessive runoff leading to flooding and (ii) erosion of the soil resource resulting from this excess runoff. However, whilst knowledge regarding soils and droughts exists in the scientific literature, a coherent understanding of the wider range of impacts has not been produced for the UK. This essay attempts to draw together knowledge from studies in the UK and abroad regarding how soils respond to drought, and importantly what and where our knowledge gaps may exist. It is not an exhaustive academic review, but it offers an informed view of key processes and characteristics of drought and its impact on soils most relevant to the UK.

The essay takes the following structure:

Section 1 defines the different types of droughts and their frequency in the UK, as well as a brief overview on the likely societal impacts that droughts place on the soil and related ecosystems. In particular, the essay focuses on ‘meteorological drought’ as this is when soils experience long dry periods followed by re-wetting. **Section 2** briefly examines the behaviour of moisture in soils and the key processes that contribute to its storage and transport. **Section 3** discusses the principal changes in the physical, chemical and biological properties of soils that result from periods of drought, and subsequent rewetting and demonstrates the extensive interactions that occur between them. **Section 4** examines the processes that are involved in the rewetting of soils and links back to **Section** 2**.** A key focus is the role of infiltration and what may occur if precipitation > infiltration. This is explored further by looking at catchment-scale soil response. **Section 5** addresses what we know regarding how soils recover after drought. **Section 6** identifies knowledge gaps whereas **Section 7** focuses on how we may improve our understanding.

Much of the information in this essay relating to the potential impacts of drought on soils, is derived from experiments where wetting and drying cycles are instigated, or climate manipulations are used. This is largely because droughts are infrequent in the UK and experiments would need to be already established to capture the impact of drought on soils. The essay reports on the large number of interactions present between soil physical, chemical, and biological properties in explaining the response to drought. Soil physical structure (essential for the infiltration of precipitation) is strongly linked to interactions between the soil matrix and soil biology. Changes in soil biogeochemical cycles are often linked to the changes in soil physical structure, such as aggregate breakdown. Key gaps in our understanding are related to the resilience, response, and recovery of soil systems. Information is required regarding soil biological communities, soil water repellency and how shallow ground water may buffer the soil moisture system.

In addition, further gaps in our knowledge exist regarding multiple stressors (heat, moisture) and the impacts of successive extreme events on soil systems (e.g. drought, flood).

1. The context of drought and its impact on soils in the UK

1.1. Droughts and their termination in the UK

Various definitions of drought exist, with the simplest being the 'absence of water', or as, 'a prolonged period of abnormally low rainfall, leading to a shortage of water' [Oxford Dictionary]. However, drought is a complex, natural phenomenon, with large variations in extent, duration, intensity, and impact. The NOAA recognise several drought types including, 1) meteorological drought, 2) hydrological drought, 3) agricultural / ecosystem drought, and 4) socioeconomic drought. The NOAA proposes that 'Meteorological drought' occurs when dry weather patterns dominate an area. Hydrological drought occurs when low water supply becomes evident, especially in streams, reservoirs, and groundwater levels, usually after many months of meteorological drought. Agricultural drought happens when crops (or ecosystems) become affected. And socioeconomic drought relates the supply and demand of various commodities to drought."

Typically, long periods of dry weather in the UK occur when a blocking anticyclone (the so called 'Azores high') pushes Atlantic anti-cyclones to the north-west. Sometimes, if there is drought in southern Britain, the north-west of the country may receive greater precipitation. In the UK, a drought is usually defined as an extended period of weather (~3 weeks) when less than a third of the expected precipitation occurs. An 'absolute drought is considered 'as a period of at least 15 consecutive days when < 0.2 mm of rainfall occurs. With 'hydrological' or 'agricultural' drought there is often soil moisture present but plants can't access it, either because it is frozen or because high temperatures determine that the rate of evapotranspiration exceeds the rate of water uptake by the plant. In the UK, hydrological droughts may occur over several years, and often involve dry winters, when precipitation does not fully replenish soil moisture storage, or reservoir and water table levels have not risen in response to summer evaporation. Major droughts in England and Wales between 1800 and 2006 have been identified and these are listed in Table 1. The comments demonstrate the complexity of drought types and periods. Kendon et al. (2013) suggest that whilst major droughts in the UK such as those cited in Table 1 are infrequent, in recent decades there have been a cluster of major rainfall deficiencies (e.g. 2010-12, 2004-2006, 2003, 1995-97, 1990-1992, 1988-89).

Table 1: Major droughts in England and Wales 1850 – present (From Marsh et al. 2007; Kendon et al. 2012; Marsh, 2003)

Year	Duration	Comments
1854-1860	Long Drought	Sequence of dry winters in both southern and northern England
1887-1888	Late winter 1887- summer 1888	Major drought across British Isles. Extremely dry 5 month sequence in 1887. Primarily a surface water drought
1890-1910	Long drought	Long duration with some wet interludes (e.g. 1903). Drought initiated by a series of dry winters
1921-22	Autumn 1920-22	Major drought. Second lowest 6 month and 3rd lowest 12 month rainfall totals for E & W.
1933-34	Autumn 1932-Autumn 1934	Major drought, severe across southern Britain
1959	Feb-Nov	Major drought over three seasons most severe in eastern, central and NE England
1976	May 1975-1976	Major drought. Lowest 16 month rainfall in E & W. Extreme in summer 1976
1990-92	Spring 1990 – summer 1992	Widespread and protracted rainfall deficiencies
1995-1997	Spring 1995 – summer 1997	Third lowest 18 month rainfall total for England and Wales
2003	Spring to Autumn 2003	Feb – Apr 2003 rainfall total for UK as a whole lowest since 1956, Brief interlude late spring to early summer when rainfall at near average but onset of heatwave in July signalled a very arid phase lasting to October causing an agricultural drought. Only 25-40% of average rainfall in period Feb – Oct.
2010-2012	Jan 2010 – March 2012	One of the longest prolonged droughts of the century which when terminated ended up in the wettest April to July, over England and Wales in 250 years.
2018	June, July, August	Agricultural drought caused by a series of heatwaves
2022	June to August	Many regions in England received less than half the average rainfall. Still ongoing and expected to continue into 2023.

The duration and spatial extent of drought determine the range and severity of impacts. In this essay, the focus is largely on ‘agricultural and ecosystem’ drought, given its frequency of occurrence and potential impact on agricultural and ecosystem processes in the UK. Whilst 'hydrological drought' may occur over multiple years, it is likely that rainfall occurs during these periods, albeit at lower than 'average' quantities, thus not impacting soils in the manner that a prolonged rain free period may.

A further important aspect of drought is how they terminate. In the UK, drought terminations can occur in any season, though they are more common through the winter than the summer half-year since rainfall is more 'effective' during winter as evapotranspiration is lower (Parry et al. 2016). Despite intense convective rainfall being primarily limited to the summer, there is no indication that rates of recovery from drought are higher in summer than winter (Parry et al. 2016). For soils, the process of drought termination is particularly important as the perceived threats of erosion and flooding may be influenced by the intensity of rainfall. The breakdown of drought can be divided into (i) the initial breakdown, (ii) the resumption of longer-term weather patterns that lead to more usual weather conditions, and (iii) the return of more typical soil moisture conditions. The initial breakdown of drought is of importance as this is when soil properties are likely to have been altered significantly by dry conditions. However, initial breakdown is likely to exhibit large scale spatial variability, and rainfall can be highly variable in intensity. Ideally, soft gentle rain at the end of the drought is preferable, but often dry periods end with large convective storms, with high intensity rainfall.

1.2. The observable drought – soil related impacts the public may recognise

It is likely that the principal soil related impact of drought in the U.K. that the public recognise, is that on vegetation. The dieback of vegetation during drought periods is often demonstrated in the media by satellite images of the country before and after drought (Figure 1). Along with the die back of vegetation, frequently asked questions are raised relating to crop yields and the impact of drought on food production and likely increases in retail prices (Geng et al. 2015). For agricultural production, the extent of impacts on crop yield may partially be driven by antecedent weather conditions. For example, one of the impacts of the 2018 heatwave/drought in the UK, was that mean winter wheat yields across the country were reported to have been reduced by an average of 5.1 % (Clarke et al. 2021). However, this drought followed a wet spring which boosted soil moisture, meaning that impacts on yield were less than expected. This type of scenario adds complexity to explaining the impact of drought on crop yields. Besides crops, multi-year drought can also cause tree / forest mortality. Predicting the resilience and recovery of trees and forests to drought is hindered by the predisposition of trees entering the drought event. This includes the cumulative effects of previous water deficits on combination with other environmental factors (e.g. pest outbreaks) that influence survival, mortality and recovery of tress from drought (e.g. Anderegg et al. 2007; Choat et al. 2018). A further feature of many droughts is an increase in wildfires (e.g. the Saddle Moor fires in 2018) which removes the vegetative cover, leaving bare soils, prone to soil erosion

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Figure 1: Before and after satellite photos of the impact of drought on the browning of vegetation across the UK in 2022.

A second observable factor of drought on soils is that of soil cracking. This occurs as smectitic clay minerals dehydrate and shrink. This can lead to the undermining of foundations of houses and infrastructure. The process is of major economic consequence with damage to infrastructure in the UK estimated to be ~£100 million a year, and sometimes reaching £400 million in very dry years (Harrison et al. 2012).

1.3. Soils and geology of England and their general susceptibility to drought

The impacts of drought may vary in their timing and severity as the way soils will respond will be largely determined by the properties of the soil and the parent material (bedrock or superficial geology) from which it is derived. The Soil Survey of England and Wales have identified >700 soil series and created 296 geographic soil associations across England and Wales with 92 occurring in Wales. In Scotland around 600 different soils have been identified (Dobbie et al. 2011). These soils demonstrate a huge range of properties (Avery, 1990), and also indicate the complexity of possible responses of soils to extended drying and rewetting. In addition, land-cover (vegetation type and soil sealing through construction) will also be key features determining the impacts of drought on soil moisture levels and run-off.

2. Key soil processes and properties relating to soil moisture storage and infiltration

To understand the impacts of drought on soils it is necessary to understand the key processes associated with water movement and storage in soils. This section provides a brief overview to (i) soil moisture storage, (ii) soil structure and (iii) infiltration.

2.1. Soil moisture retention – an overview

In terms of the global hydrological cycle, the overall quantity of water in soil is small, ~0.05%. However, it's importance in both the global energy balance and the supply of water to plants far outweighs its physical quantity. The water retained in soils, termed soil moisture, is the life blood of terrestrial ecosystems. Plants take up water containing nutrients from soil. The water is used in photosynthesis for growth and the production of food, feed and fibre. Moreover, soil moisture is required by the soil microbiome to function, its activity being closely controlled by soil moisture levels. In turn, the microbiome is responsible for processing dead organic matter, contributing to the global carbon and nitrogen cycles, and producing materials that are important in developing physical characteristics of soil such as its strength and structure. Soil structure is fundamental to how precipitation, landing at the earth's surface, is partitioned into that which infiltrates, and that which runs off. In addition, scientists are beginning to understand the role that soil moisture plays in determining the magnitude and persistence of heat waves through land atmosphere interactions (Miralles et al. 2019). In addition, the drying out of soils during heatwaves can play a role in mitigating deadly heat stress by reducing humidity (Wouters et al. 2022). As a result of the importance of soil moisture to earth system processes, droughts or floods which push soil moisture levels beyond their normal operating range, are likely to have a cascade of impacts.

To develop an understanding of the impacts of drought on soils, an overview of soil hydrological processes, both at the scale of the soil profile, and at the grain scale where water interacts with the soil matrix is required. The main processes that determine the storage and movement of water in soils are summarised in Figure 2. Water arriving at the ground surface as precipitation will infiltrate or run off. The water that infiltrates, distributes itself through the soil profile. The moisture stored within the plant root zone, is potentially available to plants through uptake and may be transpired back to the atmosphere by plants. Some soil moisture will be directly evaporated back to the atmosphere and never make it into the transpirable soil moisture pool. Conversely, water moving beyond the root zone may move downwards through the vadose zone and into the saturated zone providing recharge to ground water. Water may also move laterally generating interflow through the soil.

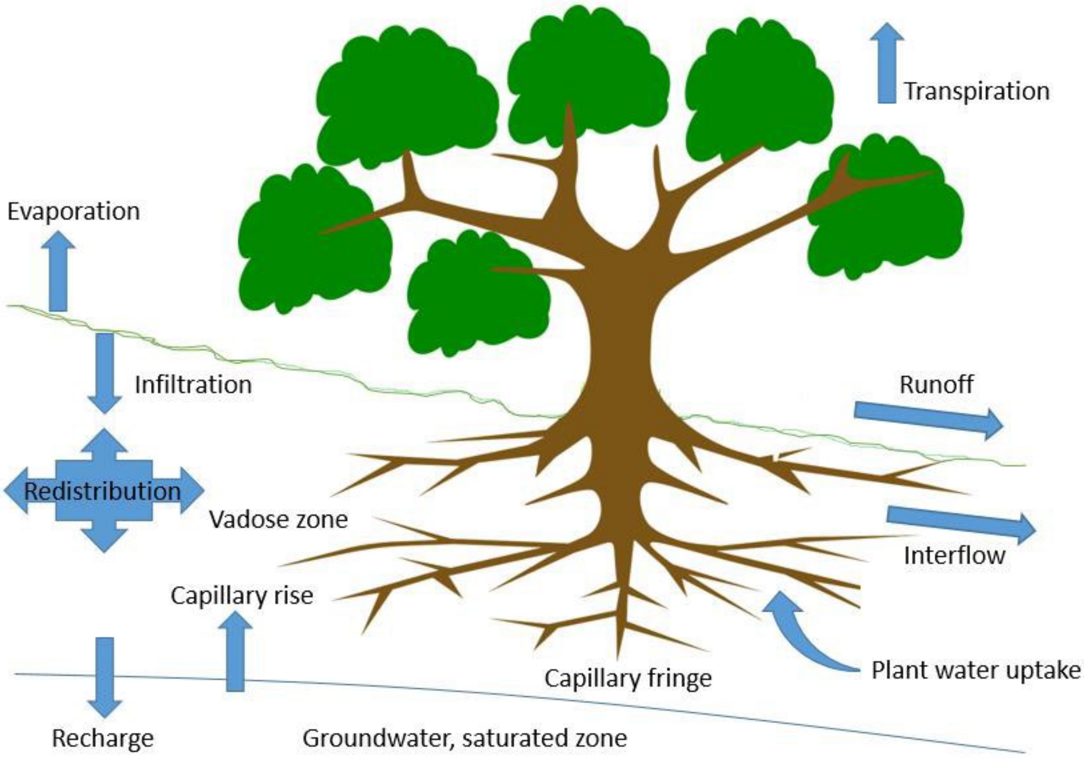


Figure 2. The basic processes in the soil water balance. Produced by David Robinson (UKCEH) for the purposes of the project

Once in the soil, water interacts with the soil matrix at the grain scale. Two important characteristics play fundamental roles: cohesion and adhesion. Cohesive forces arise from the intrinsic properties of water, in particular hydrogen bonding, which causes molecules to be attracted to one another. Cohesive forces are responsible for the phenomena of surface tension. This is the tendency of a liquid surface to resist rupture when placed under tension. It is this phenomenon that causes water to curve or bead in contact with air. At the air water interface, water molecules in contact with air at the surface have fewer water molecules to interact with and as a result form stronger attractions. This water forms spherical droplets and can support small objects such as pond skaters. Adhesive forces are those that form between the water and other parts of the soil matrix, such as clay minerals. Adhesion and cohesion lead to the phenomena of capillarity. Examples of capillarity are (i) when a narrow tube is placed upright in water and the water is seen to climb up the tube against gravity because of the strong attraction between water and glass and (ii) when a dry paper towel is dipped into water and moisture climbs up the paper towel. In soils it is the balance of these and other forces (e.g. gravity) that control the storage and movement of water.

Every soil has a specific capacity to retain water as soil moisture, based on properties such as its texture (the proportions of sand, silt and clay), organic matter content and mineralogy. The arrangement of these materials provides the soil structure and the pore network which form spaces or pores in the soil. The surface area of the materials and the size of pores they form determine the extent to which the water can interact with the material through adhesion. These forces, termed the matric forces, resist the action of gravity on the water retaining it in the soil. The matric forces are much stronger than gravity and are therefore dominant. This results in perhaps one of the most important relationships in soil science, the water retention curve. The suction (-m) is typically related to soil moisture (cm3/cm3) via a ‘Soil Moisture Release Curve’ (SMRC), with curve parameters derived from the fitting of a function (Van Genuchten, 1980) or similar equation (Figure 3).

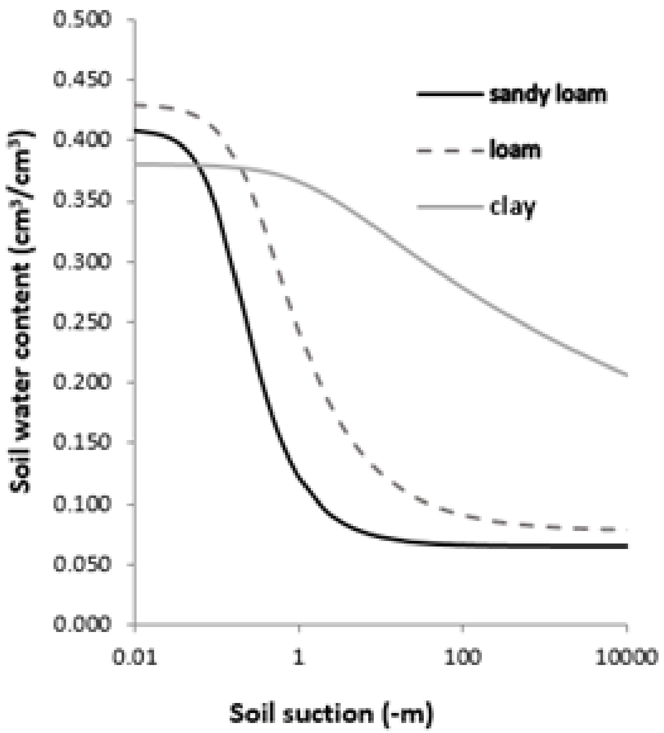


Figure 3. Soil water release curves for different soil textures. Produced by David Robinson (UKCEH) for the purposes of the project

A key characteristic of the SMRC is that when drying occurs, the large pores empty first, leaving the remaining water in smaller pores with greater interactions with soil matrix surfaces, where it is held more tightly. Moreover, Figure 3 illustrates the importance of soil texture and how fine-grained soils with greater proportions of clay, a greater surface area and small pores retain more water. Field capacity (-3.3 m, Figure 3) represents the amount of water held by the soil after excess water has drained via gravity, typically a few days after saturation.

The permanent wilting point is the suction at which plants cannot obtain water and hence wilt; the value is typically -150 m but will depend on species. Plants have evolved different water management strategies. Isohydric plants maintain a constant midday leaf water potential whereas anisohydric plants have more variable leaf water potentials, thus keeping their stomata open for longer under drought conditions, often drawing water from soil at very low water potentials (Maherali et al., 2004). Practically it is important to know that the moisture release curve differs depending on whether the soil is wetting or drying, a phenomenon known as 'hysteresis'. Thus, the equilibrium soil moisture at a given suction is generally greater in drying than in wetting. This hysteresis effect can be attributed to (i) the geometric nonuniformity of the individual pores, (ii) the contact angle effect, (iii) entrapped air which decreases the water content of newly wetted soil and (iv) swelling, shrinking or aging phenomena resulting in changes in soil structure, pore space and connectivity.

2.2. Soil structure

Soil structure is a key property in the transport of soil moisture, both downwards through infiltration and upwards through evapotranspiration processes. It also contributes to the shape of the soil moisture release curve, particularly at the wetter end of the curve, where free water draining under gravity is of greater importance. Soil structure exists at different scales, and is largely dependent on the textual, and mineralogical (e.g. clay types and contents and oxides) properties of the soils derived from the parent material, along with the climatic influences on weathering processes and vegetation type and decomposition. The term soil structure refers to the shape, size and degree of development of aggregation, the process whereby primary soil particles form into natural or artificial structural units (peds, clods, artificial or natural fragments). These form together to produce the arrangement of solids and spaces within soils and will help determine infiltration. Gao et al. (2023) suggested that the movement of soil moisture is largely via preferential flow, largely produced by a variety of biological activities.

2.2.1. Macro-scale structure

Peds and fragments come in various shapes and sizes and include shapes described as platy, prismatic angular blocky, subangular blocky and granular. Size ranges from fine (< 2 mm) to very coarse (>10 mm). These properties combine to form the macro-structure of soils and represent how the soils may break apart on a large scale and provide macro-flow or preferential flow paths. The key to how soils structure is impacted by drought relates to the depth that changes occur and how this may increase the size of preferential pathways for infiltration (Gao et al. 2022). This will be particularly relevant for clay soils, especially those that have a high proportion of smectitic clay minerals, which through shrink swell processes may form large peds, increasing preferential flow pathways (see Section 3.1). Soils forming from sandstones and siltstone lithology may have very little structure in the subsoil. For soils formed from depositional processes it is likely that a more layered subsoil may occur where different sediments with different properties lay over each other, creating differences in potential soil moisture transport.

2.2.2. Micro-scale structure

Soil aggregation is a key structural property of soils and is important in determining porosity, infiltration and gaseous exchange. It is often used as a 'soil quality' indicator. Aggregate stability measurements provide an indication of a soils ability to resist disintegration under physical stresses such as tillage, raindrop impact and wind erosion. It is often described using a 'Mean Weight Diameter' index (MWD) that characterises the structure of the soil by integrating the aggregate size class distribution into one number. However, measurements of aggregate stability are typically undertaken on specific class sizes; those classified as macro- (>250 μm) or micro- (<250 μm) aggregates (Six et al. 2000). The difference between the MWD size distribution before and after testing for aggregate stability has been termed the 'Disaggregation Reduction' (e.g. Rawlins et al. 2015). Soil aggregation is described in terms of hierarchy and is dependent on soil type. Oades and Waters (1991) examined the fractionation of aggregates after disaggregation treatments. They suggested that in alfisol and mollisol soils, macro aggregates > 250 μm diameter, broke down to micro-aggregates 20-250 μm in diameter, prior to particles < 20 μm being released. In oxisols (typically highly weathered soils), aggregation has a greater dependence on sesquioxides as soil organic carbon (SOC) is often lower. Aggregates were found to resist disaggregation initially, before breaking down into particles, thus showing no hierarchy.

The role of SOC in the formation of aggregates has been found to be a key variable determining aggregate stability (Oades, 1993). At larger scales plant roots and fungal hyphae are found to create sticky substances that enmesh particles forming aggregates. At smaller scales mucilages from roots, hyphae and from bacteria and earthworms are involved in creating smaller aggregate sizes. However, these stabilising agents can be decomposed by microflora and fauna leading to a breakdown of structural stability. The continual process of formation and breakdown of aggregates determines a soils health and drainage properties. Findings suggest that macro-aggregates often act as protection for microaggregates and the breakdown of macroaggregates normally leads to the formation of more stable microaggregates. These are generally enriched in OC and act as an important SOC store (Angers et al. 2022; Six et al. 1999, 2000). In temperate climates the stability of aggregates is strongly related to organic carbon status of the soils, texture, multivalent cations that act as bridges between organic colloids and clay, mineralogy and land use (Oades, 1984). The management of soils can impact aggregation. Soil tillage has been found to reduce fungal hyphae networks and have a significant effect on the creation and stability of aggregates (Duchicela et al. 2013).

2.3. Infiltration

Imagine yourself in the kitchen mopping up water that has spilled on the counter-top. You reach for the dry kitchen paper to mop up the water. The dry paper towel quickly absorbs the water, but its ability to soak up more water reduces as it becomes wetter. The ability of the dry paper towel to quickly absorb the water is due to capillarity, the process previously described. When the towel is dry the matrix suction in the towel is at its highest and it quickly wicks in water, slowing as the capacity of the towel decreases. Soils behave in a similar way with two components governing the ingress of water into the soil. When a wetting fluid, such as water, has contact with dry soil, the water is drawn in simply due to capillarity. When this process occurs without the influence of gravity e.g. as horizontal flow, it is considered a ‘sorption’ process; controlled by the characteristics of the soil and matrix forces. When gravity aids the process, it is termed infiltration; the sum of capillarity and gravity (Assouline, 2013). The infiltration capacity, or infiltrability is the maximum rate at which the soil can absorb water at the surface (Horton, 1940). In the case of dry soils this rate is initially higher than in wet soils, but the two reach an equilibrium and a steady state which in theory occurs at the saturated hydraulic conductivity (Ks). Based on theory, soils that have experienced drought and are dry should exhibit the highest infiltration capacity initially as shown by the solid black line in Figure 4.

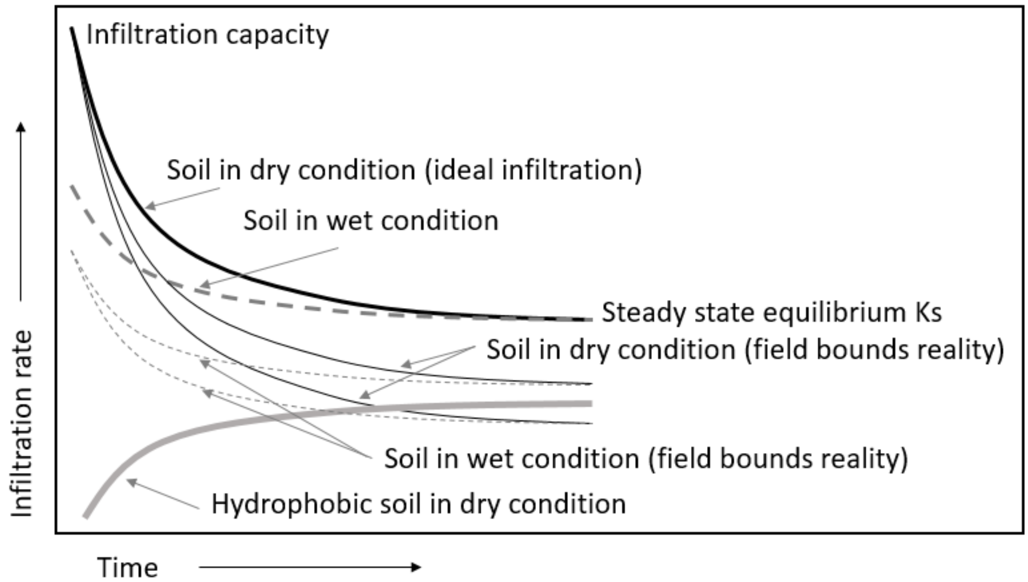


Figure 4: Diagram showing the how infiltration rate changes with time and with soil condition. Produced by David Robinson (UKCEH) for the purposes of the project.

3. What happens to soils when they experience extreme drying and later rewetting

In this section the major changes that soils experience when exposed to long periods of drying, such as those produced by an 'agricultural or ecosystem' drought, are examined. Importantly these changes may impact soil moisture movement and infiltration.

3.1. Physical changes in soil structure in response to drought and rewetting

This section considers the physical behaviour and structure of soils in response to drying and re-wetting and the likely impact of drought conditions. The processes described are important as the distribution and spatial extent of different soil types in catchments of soils prone to these physical changes will determine the catchment response when the drought terminates.

3.1.1. Shrink-swell and cracking

The process of shrink-swell and soil cracking is a common feature in many clay soils, particularly across southern Britain, and is the process known to potentially cause significant damage to infrastructure. This process occurs in soils that contain smectitic clay minerals which demonstrate change in volume in response to its moisture content. Soils susceptible to shrink swell behaviour are shown in Figure 5. Smectitic or expandable clay minerals have an interlayer space in their structure, and may swell up to 10% when wet by adsorbing water into the interlayer space. This structure will collapse causing shrinking when drying out. Typical soil series where this occurs in England are those such as the Denchworth, and Hanslope series. Whilst the soils are likely to crack to some extent (depending on the amount of smectitic clay available) in most summers due to drying, the prolonged drying in periods of drought, can enable large, deep cracks to form, which may extend to depths of 3 m (Harrison et al. 2012; Hawkins, 2013; Jones et al. 2020). Infiltration has been examined in these soils and modelled using cores and lysimeter blocks (Jarvis & Leeds-Harrison, 1987; Bradley et al. 2005; Harris and Catt, 1999). The hydraulic conductivity in these soils when there is no sign of cracking was found to be as low as 0.26 mm to 4.38 mm h-1 in an Evesham clay soil (Jarvis & Leeds-Harrison, 1987), whilst hydraulic conductivity at 0.5 m depth was as low as 0.4 mm h-1 (Leeds Harrison et al. 1986). However, in a small, replicated field experiment on Denchworth soils, where cracking had appeared, typically over 90% of the runoff was found to move rapidly through the subsoil into drain flow within about 2 hours of peak rainfall (Harris and Catt, 1999). This process is known as preferential flow. The rewetting of the soils in autumn is likely to cause some swelling and 'annealing' of cracks, although there is still debate regarding the importance of hysteresis in shrinkage and subsequent swelling processes in clay (Holtz & Kovacs, 1981). The cracking of these soils, in the context of water availability, can be considered an important 'resilience' factor during a typical spring and summer as they allow rapid recharge and re-wetting. This process is widely recognised (Somasundaram et al. 2018).

A further feature of soil cracking is that it may increase evaporation rates from soils. Poulsen (2022) suggested that in moist cracked soils, the soil cracking increased evaporation by 60-65% compared to uncracked soil for wind speeds ranging between 0-5 m s-1 and a range of cracking densities. Hatano et al. (1988), working in cracked clay soils, suggested that evaporation from the soil surface accounted for 2-12% of the evapotranspiration budget, but evaporation from the cracks alone accounted for 10-50% of the evaporation, although the cracks accounted for only 6.7% of the soil surface area. Thus, in cracking clay soils, drought may increase and enlarge soil macropore size and evaporation surface, accelerate the evaporation of pore water, lower the water retention capacity of soil and degrade physical and mechanical properties (Zeng et al. 2020). Soils with different SOC contents with depth can have different shrink-swell and infiltration behaviour between soil horizons.

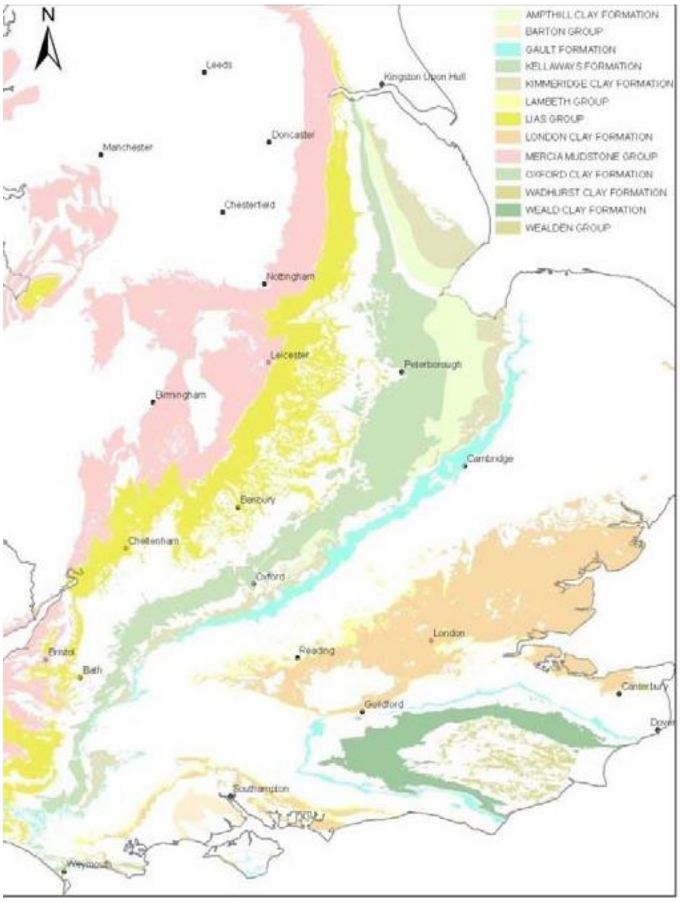


Figure 5: Soil Parent Materials in England which may exhibit shrink swell characteristics due to smectitic clay concentrations. Contains BGS Geology Data©UKRI and OS data © Crown copyright and database right 2022.

3.1.2. Aggregate breakdown

Some studies have examined the manipulation of climate on soil aggregate stability. Those focused on creating drought conditions and provide an indication as to how aggregates may respond to dry periods in the UK. Zhang et al. (2018) examined aggregate stability in afforested areas on a loess soil in China, where precipitation was excluded, and understory removed. Results showed that the removal of understory and the exclusion of precipitation and throughfall resulted in decreased soil aggregate stability and aggregate -associated organic C pools, whilst having no influence on total SOC. Yang et al. (2019) carried out a similar precipitation exclusion experiment in sub-tropical plantations. They found precipitation exclusion reduced soil aggregate stability as expressed by the mass fraction of macroaggregates and the MWD of aggregates. In these environments, a reduction in free Al oxides and an increase in porosity was considered the likely cause. These experiments demonstrate that soil aggregation may be impacted by changing environment, but UK studies have yet to be undertaken. However, a reduction in mucilage and polysaccharide production by roots and soil fauna in dry soils is likely to impact aggregate stability (Treseder et al. 2010).

The strength of soil aggregates will help determine a soils behaviour to drought and rewetting. Aggregate disruption may occur through (i) slaking, (ii) the differential swelling of clays, (iii) mechanical disruption through kinetic energy (e.g. more intense rainfall) and (iv) physiochemical disruption. Slaking is probably the most important and widespread mechanism and is caused by the trapping of air through rapid wetting. The trapped air then pushes soil particles apart, collapsing the particle (Xiao et al. 2017). This may lead to a decline in porosity as pores are blocked by the released particles. Splash detachment is another important process in aggregate breakdown and is affected directly by the physical properties of rain such as raindrop size, terminal velocity, rain intensity, kinetic energy, and momentum, along with soil properties such as organic matter content, calcium carbonate content, particle size and soil structure (Sharma et al. 1991). Xiao et al. (2017) examined rainfall intensity (60 mm hr-1) at different heights (0, 1, 1.5. 2 and 2.5 m) to achieve different quantities of kinetic energy. The rates of splash erosion followed the order of loamy clay soil < clay loam soil < sandy loam soil, but the MWD of disintegrated aggregates followed the reverse order. A feature of slaking and splash detachment of aggregates is that it increases unattached fine particles that can be transported by splash erosion, downslope. Assuming drought reduces the strength of organic matter binding of particles, then it is likely that drought will lead to some reduction in aggregate stability with implied consequences for processes such as infiltration and erosion.

3.1.3. Surface crusting

Surface roughness is a key control on overland flow, with greater variation in surface roughness influencing the depression storage of water, infiltration, overland flow velocity and overland flow pathways (Govers et al. 2000). A further consequence of aggregate disruption, especially via intense precipitation and splash detachment is the formation of layers of fine soil particles that when dry create a layer of ‘surface sealing’. When hardened by the sun, this dry layer has the potential for (i) reducing infiltration, (ii) increasing surface flow and erosion, and (iii) impeding the shoots of plants breaking through the surface.

3.2. Biological impacts of droughts and rewetting on soils

3.2.1. Impacts of extreme drying on soil ecosystem function and nutrient cycling

The soil is an ecosystem where plant roots and the soil microbiome interact with the abiotic components of soils, developing soil structure, which helps to define a soil's specific infiltration properties. Microbial and fungal populations are instrumental in the processing of waste and the cycling of nutrients in soils, producing substances that help bind soil particles together into aggregates in addition to changing biogeochemical cycles. Thus, understanding how extended periods of drying and heat will impact plant, microbial, fungal and soil faunal functioning, and how this contributes to the infiltration of water and a range of other processes is important. Moreover, many microbial and fungal populations are involved in symbiotic relationships with plants. Thus, impacts on soil microbial and fungal populations communities and function may impact plants and vice-versa. Here we explore some of the more important responses related to drying and drought for the belowground ecosystem.

3.2.2. Soil biological responses to drought

A meta-analyses summarised the findings of drought effects on biological soil responses (Abbasi et al. 2020) which are mainly mediated by plant roots, soils fauna and soil microbes. Plants, whether grasses, herbs, agriculture crops, single trees or forests, each have unique root systems that vertically and horizontally anchor them in soil. Plant rooting patterns are species specific, and also depend on prevailing soil characteristics (e.g. Thomas 2000). Soil microbes are interconnected with plant roots, in the rhizosphere, which are the areas of highest microbial activity. It has been shown that the soil microbial community can improve the drought tolerance of plants (Kannenberg and Philips 2016, Kour and Yadav 2022, Rubin et al. 2017). In addition to the plant rhizospehere, there is microbial activity in the bulk soil which is not directly associated with plants. This activity is much lower, but it may contribute to soil moisture movement by creating biofilms. Myccorrhizal networks connect rhizospheres of mainly trees and woody species across bulk soil. Chandrasekaran (2022) for example concludes that plant root physiology changes under drought, allowing arbuscular mycorrhiza fungi (AMF) to alleviate drought stress by enhancing AMF colonization and increasing nutrient (N, P, K) uptake. Similar results suggested that soil K is a driver of changes in root morphology and the alleviation of plant drought stress (Xu et al., 2021). Belowground interactions and responses to drought in the physical soil environment influence how plants and microbes (and fauna) experience reductions in soil moisture content, the mechanisms they have as a community to deal with reduced hydrological connectivity, and which changes they may induce (Figure 6). All adaptive mechanisms in response to drought will have an effect on the soil environment which cause changes in other properties (e.g. soil C content).

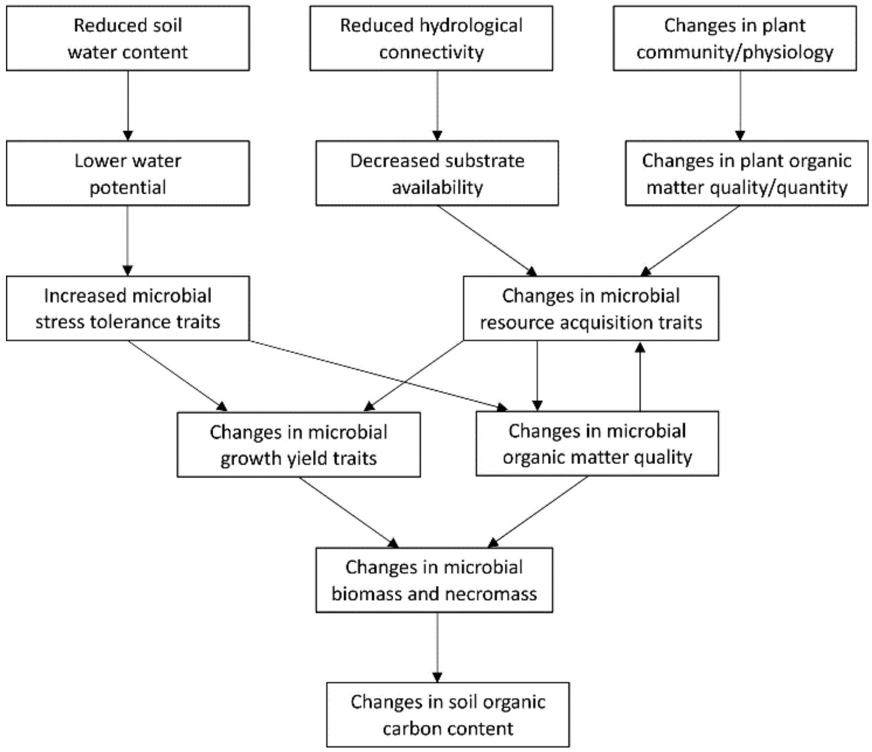


Figure 6: Theoretical framework of how drought affects soil microbes – and how this links to the soil carbon pool. Reproduced from Malik and Bouskill (2022) under the CC BY 4.0 licence.

3.2.3. Effects of drought on soil Carbon and Nitrogen processes

In general soil biological processes decline during droughts; water-logged soil being the exception. However, the belowground chemical environment may change during drought: it has been found that (i) root C concentration can increase without an increase in belowground biomass and (ii) total soil C accumulates due to overall decreased biological activity (Abbasi et al. 2020, Qu et al. 2023). The increase in root C content may be due to reduced mycorrhizal and microbial activity, trading C for N, however, the root C:N ratio may not be significantly changed. Root C may also be used to expand the root network, exploring the soil vertically for alternative water (and nutrient) sources. This is required to maintain the water potential needed for water transport within the plant; if the soil moisture falls to the matric potential of the permanent plant wilting point, water movement within the plant is disconnected and the plant dies.

Prolonged low soil moisture movement may cause reduced availability of N and macronutrients, leading to plant browning (loss of chlorophyll) and earlier litter production as defence mechanisms to survive drought fail (Bogati and Walczak 2022). During prolonged drought, loose litter may accumulate on top of the soil surface. Under average rain patterns, this litter will be decomposed by soil microbial enzymes (fungal and bacterial) which ensures that OM and nutrients are returned to the soil. During prolonged drought, lower (to none) faunal and bacterial activity and lower enzyme activities (Abbasi et al. 2020, Bogati and Walczak 2022) slow litter decomposition. This litter is more likely to be re-located by wind and this can cause a reduction in soil nutrients and OM – with OM being a critical component of soil aggregates and soil fertility (Bogati and Walczak 2022). In addition, re-located litter cannot build a barrier between the sun and the soil surface, meaning the soil may 'bake', crack or shrink earlier, leaving the soil surface exposed to erosion processes.

Land cover has been found to influence soil C and N responses to drought, although responses have been found to be variable. Soil inorganic N (NH4+ and NO3-) has been found to be higher during drought (Abbasi et al. 2020, Qu et al. 2023) whilst NH4+ and NO3- were observed to be more abundant under drought in soils with grass, but not in forests (Qu et al. 2023, Bogati and Walczak 2022). This decrease of NH4+ availability in forests under drought, suggests that forests may have greater ability than grasses to maintain the soil nutrient status under drought. This is likely due to their differences in rooting systems and overall life strategies. Additionally, labile C in the form of DOC was found to increase under drought in forest, grassland and shrublands (Bogati and Walczak 2022), making soil C loss more likely when drought terminates.

When drought terminates and soil rewets, soil biological activity can accelerate, often overshooting normal root and microbial activity until an equilibrium between the microbial community and the exchange with plant roots is reached. This initial re-wetting causes a peak of soil respiration losses and increased nutrient mineralisation (Barnard et al. 2020, Manzoni et al. 2012); if there are frequent drying and re-wetting cycles, soil C and nutrient contents may decrease. This was explored by Barnard et al. (2020) who suggest that besides climate, soil type and prevailing nutrient and C sources (as set by plants), the magnitude of change in soil water potential with the termination of the drought determines the soil C loss. The highest soil C loss has been found from high-C containing soils during drying and re-wetting cycles (Canarini et al. 2017).

3.2.4. Microbial responses

In their perspective paper, Malik and Bouskill (2022) summarised the impacts of drought on microbial traits and how this feeds back to the soil C cycle. The diversion of resources away from microbial growth during drought is proposed to change SOM chemistry, and its persistence, depending on the microbial compounds produced. Firstly, microbes must maintain a positive intracellular turgor pressure relative to their external environment. This is achieved by the synthesis or transport and subsequent accumulation of solutes within the cytoplasm. If the external solute concentration increases (hyperosmotic shock) water efflux from the cytoplasm into the surrounding cell can result in a drop in turgor pressure. In contrast, a quick decline of external osmolarity (hypoosmotic shock) may lead to a swift influx of water, and an elevation in turgor pressure. Turgor pressure is highly important for microbes which have evolved different pathways to maintain their intracellular turgor pressure under fluctuating external osmolality. Three pathways of microbial drought adaptation were described: (i) the production of extracellular polymeric substances (EPS) which are the main components of biofilms, (ii) the acquisition of solutes from the surrounding environment, but this may be limited by the constraint of diffusion in dry soils, and (iii) dormancy via spores. The energetic expenditure decreases from (i) > (ii) > (iii) and will ultimately govern the microbial response to drought (Manzoni et al. 2014).

Soil microbial activity ceases, at a nearly constant soil moisture threshold with a water potential of about -13.6 MPa in mineral soils and -36.5 MPa in surface litter (Manzoni et al. 2012). The constraints microbes experience within soils result from limited water diffusion. Or (2007) provides information on activity thresholds for different microbial species (Figure 7) with bacteria, but also fungi and soil fauna displaying species specific declines in activity with decreasing soil water potential; the order of sensitivity being soil fauna > bacteria > actinomycetes > fungi (Table 2; Manzoni et al. 2012). Manzoni et al. (2012) also found microbial activity thresholds were much lower in undisturbed soils (compared to experimentally broken up soils), likely due to intact aggregates and restrictions on diffusion occurring naturally. Interestingly, they did not find an effect of biome, climate or fungal-to-bacterial ratio on the soil water potential threshold governing activity in the soils.

Soil microbial communities change when exposed to drought, with drought-resistant communities becoming prevalent or changes occurring in soil microbial community functioning (Fuchslueger et al. 2016, Canarini et al. 2021). Canarini et al. 2021 undertook a drought experiment to test the microbial memory to drought in-situ. Plots were subjected to (i) no drought, (ii) a single drought, and (iii) 10 years of re-occurring droughts in a mountain grassland. They studied the microbial community, as well as C, N, and other nutrients. Microbial responses were measured and found to be different after 1 or 10-year droughts with soil nutrient components changed after one year, but after 10 years being similar to controls. However, although soil nutrient status was not different between controls after 10 years, the soil bacterial and fungal communities were. This finding suggests that long-term drought events change the soil microbial community to maintain ecosystem functioning.

Table 2: Data extracted from Manzoni et al. 2012

Water potential (MPa) threshold	Fauna / bacteria / fungi
-0.5	Soft-bodied fauna (e.g. nematodes)

-0.6	Nitrifying bacteria
-2	Hard-bodied fauna (e.g. arthropodes)
-2	Gram positive bacteria
-4	Wood decaying fungi
-10 to -3	Fungi
-10	Actinomycetes
-10	Threshold where solute-diffusion ceases: see Figure 7: this is where microbial activity drastically declines.
-60	Some xerophilic fungi

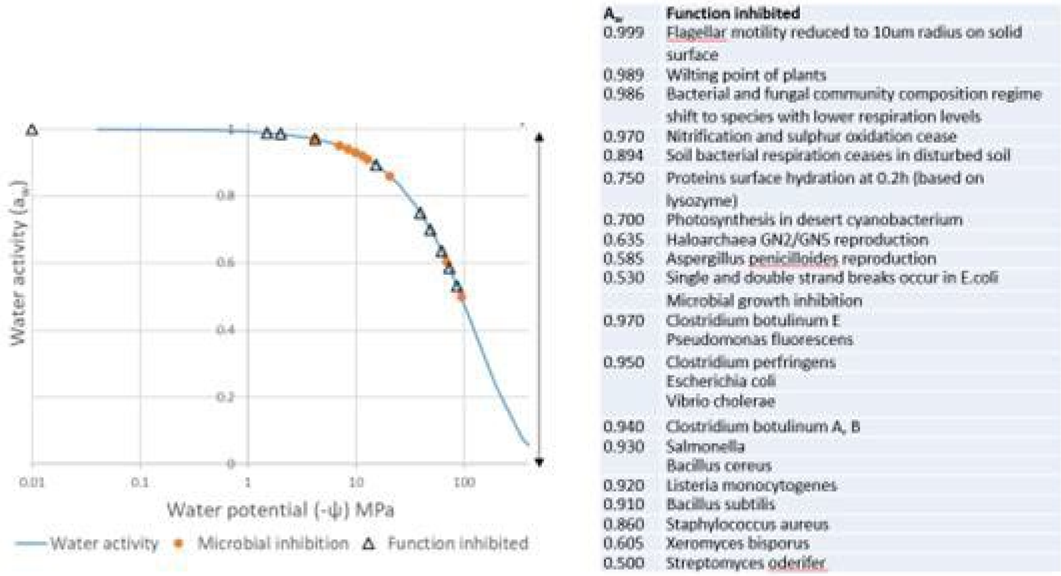


Figure 7: Relationship between water activity, water potential and the impacts on microbial function. Produced by David Robinson (UKCEH) for the purposes of the project.

3.2.5. Soil water repellency (SWR)

As outlined above, microbes adapt to drought sequentially, changing the function of different microbial communities in the soil. Some microbes are more resilient to drought than others and a range of different responses occur within different communities. An increasingly documented phenomena is the development of soil water repellency (SWR), which may influence infiltration rates. Initially, SWR was linked indirectly to drought due to its occurrence after wildfire (Doerr et al., 2000), when multiple organic compounds are generated which may coat the soil leaving it water repellent. Evidence suggests that the magnitude of SWR depends on the temperature of the wildfire burn. Compounds formed at low temperatures may melt and coat soil grains, whilst at higher temperatures compounds may be completely oxidised. A number of interlinked processes occur following drought induced wildfire, SWR being one and soil erosion often another (Doerr et al., 2009b). With climate change it is likely that the UK will experience more drying events and so interlinked wildfire, SWR and erosion are to be expected. Much may be learned from the experience in Mediterranean countries where this is more common. However, an increasing body of research suggests that SWR is not simply a phenomenon associated with wildfire. Doerr et al., (2009a) indicated that background levels of SWR in soils are often high so the relationship with wildfire is complex.

Water repellency is also related to abiotic and biotic phenomena found in a range of environments. Perhaps one of the most widely recognised occurrences is the super hydrophobicity found in some aquatic plants. Lotus leaves for example, use water repellency as a self-cleaning mechanism. In this particular case, water repellency is generated physically by fine leaf hairs, whilst many plants also have waxes on leaves to reduce water loss. Practically, many golf course managers are aware of the affects as SWR as it is a problem in turf grass causing dry-spots on golfing greens (York and Canaway, 2000). In soils, SWR is more commonly associated with organic compounds, especially waxes and long-chain fatty acids (Graber et al., 2009). However, microbes can also become, or exude water repellent compounds; golf greens again being an example where basidiomycete fungus may cause ‘fairy rings’ (York and Canaway, 2000). There is now a growing literature of different organisms or groups of organisms that under different circumstances become water repellent. While commonly associated with drought, biological SWR can also occur in response to flood.

Soil water repellency may also occur due to biological organic compounds produced because of stresses (Rillig et al., 2019), such as too much or too little water; plants, bacteria and fungi all exude water repellent compounds (Seaton et al., 2019). Their impact on infiltration will depend on the soil structure. Soils with a uniform pore size distribution will often experience reduced infiltration initially. Moreover, the repellency may also lead to enhanced air entrapment. However, in soils with large macropores, water ponded on the hydrophobic surrounding soil may infiltrate preferentially in macropores. It has been proposed that this may result in a competitive advantage for some plants in water limited environments (Robinson et al., 2010).

Stress, or environmental extremes, is an emerging theme with SWR. Rillig et al (2019) undertook an experiment examining multiple global change factors, or stressors affecting soil microbiology. An important result was that soils generally coped with one or two stresses, but when the number increased to 8 or 9, processes such as SWR occurred. Seaton et al (2019), used a national data set of SWR measurements to assess the role of stress in its occurrence. Results suggested that, ‘a wetter climate and low nutrient availability alter plant, bacterial and fungal community structure, which in turn are associated with increased soil water repellency across a large-scale gradient of soil, vegetation and land-use.’ In the desert in Utah, Robinson et al (2010) proposed that stress through extreme drying, led to SWR that provided a soil moisture maintenance pool feedback process for woody vegetation. This is an emerging research area, but Seaton et al. (2019) found that 92% of soils sampled in Wales showed signs of SWR, indicating that its occurrence is the norm and not the exception and one that drought conditions in soil may promote. The advantages of water repellency to micro-organisms involves water regulation and management. This may involve keeping water away under saturated conditions and creating air pockets, as observed with fungi. Conversely it may involve keeping water in the organism as with fungal hyphae during drought (Unestam, 1991). Moreover, water repellent films have been shown to develop as biofilms protecting organisms (Epstein, 2011), including from rapid rewetting following drought. These factors, and potential benefits to plants under drought conditions indicate natures adaptations, that to date have not been fully explored.

3.2.6. Macrofauna

Soil macroinvertebrates are organisms with body widths generally between 2 and 30 mm and in temperate systems they can account for up to 79% of soil animal biomass (Gongalsky 2021). In turn, these invertebrates play an important role in many soils by driving dynamic changes in biophysical properties, with several groups being characterised as soil ecosystem engineers (Jouquet et al. 2006). They include common taxa such as earthworms, woodlice, and millipedes, as well as the larvae of insects. The movement of earthworms through soil shapes the architecture of the matrix, particularly the arrangement and size distribution of the pore network (e.g. Capowiez et al. 2014; Capowiez et al. 2015). Some earthworm taxa generate characteristic biogenic structures such as deep vertical burrows and surface casts. The feeding activities of several soil macroinvertebrate taxa (earthworms, woodlice, millipedes) involves the breakdown of litter material and the distribution of faeces, thus influencing the incorporation of organic matter into the mineral soil. Together, these changes in soil architecture and biophysical properties affected by soil macroinvertebrates have significant impacts on soil hydrology and the ability of soil to retain moisture.

Earthworms, being soft-bodied and reliant on moist conditions to allow oxygen exchange through their skin, are not tolerant of dry conditions. It is typical that their behaviour changes to avoid drier summer soil conditions of a temperate climate. Earthworms may migrate down the soil profile to seek more moist conditions and escape desiccation. Some species may enter a dormant or semi-dormant state to conserve moisture and energy - aestivation, a state of diapause during dry summer conditions, is driven by drought conditions. Earthworms create an aestivation chamber, surrounding themselves in mucus and knotting their body. Laboratory experiments by Holmstrup et al. (2001) with an endogeic earthworm species, Aporrectodea calignosa, and a two week drought, found diapause to be induced below -20 kPa water potential, and all earthworms entered diapause beyond -40 kPa. It was also noted that these modest water potentials inducing diapause may be linked to the abrupt nature of the change under experimental drought conditions, whereas earlier field studies under progressive drought found diapause to be induced at around -160 kPa (Holmstrup et al. 2001). The increasing number of earthworms entering diapause as drought stress progresses from -0.061 MPa to -0.19 MPa over three weeks (McDaniel et al. 2013) supports this.

Drought also has the potential to alter earthworm populations and community composition over time through differential impacts on species and functional groups. Eggleton et al. (2009) studied earthworm populations in a UK pasture woodland over 72 consecutive months, with the sampling period including two dry phases. Seasonal patterns were evident across most of the common species, with maxima in winter and minima in summer, and this was reflected in the total earthworm abundance (Eggleton et al. 2009). However, severe declines were noted for the surface dwelling epigeic, Dendrobaena octaedra, following successive dry summers, and no adult earthworms were found at all in September of 2003 and 2006 following the dry phases. A review of climate change effects on earthworms by Singh et al. (2019) suggested that, while drought generally had negative impacts on earthworm populations, more evidence is needed on the interaction with other context-dependent factors such as soil type and environmental stresses. Prolonged drought may increase earthworm mortality. McDaniel et al. (2013) demonstrated that a 3-week drought stress, resulting in -0.19 MPa water potential, led to 14% mortality of the endogeic A. caliginosa. Evidence also suggests, however, that earthworm populations have the ability to recover from severe drought events. In the study of Eggleton et al. (2009) populations of all affected species recovered from successive dry phases after only one year. Both behavioural (i.e. aestivation) and life-history traits (i.e. cocoon production) likely contribute to this recovery. It took only 74 days for earthworm cocoon production to recover in treatments where earthworms had been induced to diapause (Holmstrup et al. 2001). It is not well understood how the impacts of drought on earthworm communities, and the transition to recovery, feedback to modulate their dynamic impacts on hydrological and biogeochemical processes in soils.

3.3. Impacts on Drought and rewetting on soil chemistry

Two major ways that drought impacts soil chemistry are (i) the impact on major nutrient cycles and (ii) impacts caused by the drying of normally wet organic soils. The perturbations caused by drought may result in changes in the transport of macro-nutrients and contaminants to water and air. These two areas are discussed below.

3.3.1. Impacts on macro-nutrient cycling in soils

The major influences on macro-nutrient cycling in soils are intimately linked to the physical and biological responses to drought previously discussed. Patel et al. (2021) recognised that a soils texture and porosity were important factors in the response to drought and flooding as they control microbial communities access to organic substrates and thus the chemical response. Whilst a major impact of drought may be the decrease in the microbial activity and the recycling of nutrients, He and Dijkstra (2014) suggest that the availability of water may be the main diver for reduced plant growth, rather than the availability of N and P, which may increase in some ecosystems (see section 3.2.4).

Soil moisture is important as plants receive nutrients through the pore water which decrease with soils drying out (da Silva et al. 2009; He & Dijkstra, 2014, Yue et al. 2018). It has been suggested that decreased P uptake by plants may be associated with the death of surface roots, the reduced soil hydraulic conductivity increasing root soil air distance and increased root tortuosity reducing flow transmission (Suriyagoda et al. 2014). Bi et al. (2023) reviewed the impacts of drought on soil phosphorus in soil-plant systems. Some studies found an increase in soil P availability due to drought. Possible explanations suggested are that during droughts, microorganisms die and release P, whilst the uptake of P by plants decreases (Yue et al. 2018). It has also been suggested that the activities of plant phosphorus-acquisition enzymes decrease in the mid-to late period of drought (Gao et al. 2021). Turner and Haygarth (2003) examined the response of P in drying soils from permanent lowland pastures in England and Wales. They suggested that increases in soil available P after drying was linked to (i) enhanced physical weathering and the disruption of organic matter coatings on clay and mineral surfaces, revealing previously protected surfaces where P sorbs, (ii) the release of soluble P due to the lysing of microbial cells and (iii) large soil aggregates breaking down, reducing SOM protection and increasing soil surface area for P desorption. Increases in N availability may also occur for similar reasons. Due to these processes, an enhanced pool of nutrient concentrations is often available at the termination of drought. These may be susceptible to transport to water courses through leaching (N) or associated with eroded sediment (P). Some soil systems may be more susceptible to enhanced transport processes. Increased leaching may occur when there is the presence of excess N and cracked soils leading to greater conductivity upon rewetting to the water table (Kalus et al. 2020). Many clay soils that crack also have under-field drainage systems and the combination of soil cracking and drainage systems may act as a rapid transport system of macro-nutrients to surface waters.

As drought terminates greenhouse gas emissions may increase. This is typical of wetting and drying cycles in soils and is often called the ‘Birch effect’ (Birch, 1958). Whilst drought causes a decrease in microbial activity and soil respiration, rewetting will stimulate microbial populations, leading to an enhanced pulse of CO2, CH4 and N2O. The processes driving this are (i) a decrease in the physical protection of SOM caused by the disruption of aggregates, thus creating a pool of nutrients to be exploited and, (ii) a physiological effect where compatible solutes produced by microorganisms and accumulated in cells to maintain the balance of osmotic pressure under drought conditions will be disposed of through the rupture of cell membranes. Again, this provides a food source for soil microbes and fauna. These effects are likely to be variable between land-cover types. However, despite this pulse being produced on rewetting, studies have suggested that the amounts of GHG’s being released may decrease overall because of the impacts of drought over its period. Jin et al. (2023) carried out a meta-analysis and found (i) drying-rewetting cycles decreased CO2 emissions by an average of 9.7% and did not affect emissions and uptake of soil CH4 and N2O; (ii) the impacts of drought and rewetting were different between ecosystem types. They suggested that the impacts of drying-rewetting was positively regulated by organic C and N concentrations, pH, clay concentration, and soil depth for CO_2 emissions, and negatively controlled by C:N ratio and silt concentration. For N_2O emissions, negative controls were C:N ratios, silt concentrations and soil depth. Overall results suggested that CO_2 emissions were significantly decreased by drying and wetting cycles while the fluxes of CH_4 and N_2O were not affected. Results, especially for CO_2 suggest that the pulse created on re-wetting could not compensate for the deficiency of CO_2 emissions during the drought period, and that the substrate created on re-wetting is quickly utilised.

The temperature response of soil CO_2 emissions was largely unaltered with (experimental) warming across most ecosystems (Carey et al. 2016). This follows the normal temperature sensitivity of soil processes where, on average, with every 10oC increase in temperature, soil respiration increases 2.6-3.3 times (Bond-Lamberty & Thomson 2010, García-Palacios et al. 2021). Temperature-induced increases in soil CO_2 emissions requires soil activity (plants and roots) as well as available C. Available C will partly derive from plant root exudates, but also from increased mineralisation of SOM, and thus, the reduction of the soil C pool (García-Palacios et al. 2021). The highest loss of soil C with warming is observed from high-C soils (Crowther et al. 2016).

3.3.2. Impacts on wetland soils

The drying out of wetland (organo-mineral, peat) soils caused by extended dry periods and drought may lead to potential changes in soil chemistry and their vulnerability (Stirling et al. 2020). A key mechanism when wetland and peatland soils dry, is to allow ingress of oxygen into the soil profile, producing an increase in the oxidation of organic matter and reduced inorganic species, particularly sulphides. Drying of many wetland soils often results in cracking which aids the ingress of O_2 to greater depths, enhancing the changes to biogeochemical cycles (Estop-Aragonés et al. 2012). Oxidation of sulphides can cause soil acidification and metal mobilization which may impact surface water quality. Increased oxygen can also impact GHG production, with decreases in the production of CH_4 through oxygenation, although the degassing of the profile may increase emissions (Estop-Aragonés et al. 2016). Increases in the production in N_2O may also occur through increased O_2 availability and an increase in nitrate caused due to ammonium oxidation (nitrification) (Chen et al. 2012). Leaching of this nitrate into reduced zones can lead to increased N_2O production. Drought typically increases CO_2 production in previously wet soils due to increased decomposition of OM under aerobic conditions. Increasing wetland soil temperature during drought, while soils maintain moisture and substrate availability can also increase OM decomposition due to decreased effort being required to reach activation energy for chemical reactions (Moinet et al. 2018). There are as ever differences produced in different organo-mineral soils and peat soils because of their properties (Stirling et al. 2020). Hughes et al. (1997) investigated summer drought in a small area of flush wetland in a three-year experiment in Wales which included summer drought / rewetting cycles. They found drought impacts on peat-water chemistry included; (i) a decrease in the summer peaks of DOC and Fe concentrations in soil solution and (ii) a decrease in acidity and (iii) an increase in the natural autumn-summer peaks in sulphate and acidity concentrations in the peat water. Similar, increases in sulphate concentrations were later found after a natural drought. The increase in sulphate is considered due to the oxidation of reduced S compounds in the peat. This has implications for contaminated wetland and peat soils in old mining areas of the UK as the decrease in pH due to sulphide oxidation could mobilise many metals (e.g. Cd, Pb, Zn).

4. What happens to soils and hydrological processes as catchments re-wet

The key process involved in the rewetting of soils is that of infiltration. If infiltration exceeds precipitation, it is likely that runoff and erosion will be low. Infiltration rates may be influenced by the changes in soil structural and biological properties and processes described in Section 3. This section firstly examines the abiotic and biotic factors that may influence individual soil profiles types and then secondly, these impacts are examined in the context of how they may impact catchment response.

4.1. Impacts on soil infiltration upon rewetting

4.1.1. Abiotic factors influencing infiltration

In Section 2.3 the process of infiltration is described. A key theoretical point is that dry soils should have the greatest potential for infiltration. However, as has been demonstrated, many soil processes associated with drought and rewetting may influence the infiltration process, making predictions of the rate of wetting more difficult. Thus, even, when the precipitation rate does not exceed the infiltration capacity, soils will wet, but not necessarily as might be predicted from laboratory measurements. A key element of soil hydrology is the ability to predict the time to ponding and when run-off may start. Infiltration in the field and its steady state hydraulic conductivity (K) can vary substantially from the ideal steady state infiltration rate represented by the saturated hydraulic conductivity (Ks) determined from laboratory measurements. The following are some of the key factors that are likely to alter the ideal steady state infiltration rate and so impact wetting when drought terminates:

Storm structure: As discussed, intense storms may generate rainfall that exceeds infiltration capacity. However, if the infiltration capacity has not exceeded the temporal variability of the intensity within, the same rainfall event can induce drainage and redistribution to occur simultaneously in the same soil profile (Assouline, 2013). As a result, hysteresis in the relationship between suction and water content can affect the infiltration process.

Air entrapment: When water infiltrates into soils it rarely fully wets the soil. Air becomes entrapped in the soil and reduces the steady state infiltration rate as a result. Measurements of steady state infiltration rates in the field compared to those produced for saturated hydraulic conductivity in the laboratory are typically one half to two thirds in magnitude (Figure 4; Field bounds reality) (Miyazaki, 2005). The soil water content that the soil saturates at with air entrapment, which is lower than the true saturated soil water content, is often called the satiated water content. This satiated water content is often ~5% lower than the true saturated value.

Frost and thermal effects: If the soil experiences freezing conditions ice crystals can form in the soil and have the same impact as entrapped air, reducing the pore space available for infiltrating water and hence reduce the hydraulic conductivity. This will only be an issue early in the year when freezing conditions follow winter or spring drought. However, thermal effects more widely, can alter the hydraulic properties by changing the surface tension and viscosity of liquid, or by altering the soil structure due to thermal expansion of minerals. The former is considered the main influence and differences are likely to be less than an order of magnitude (Gao and Shao, 2015).

Soil structure and sealed layers: Dry conditions may reduce the cohesion between soil particles, making them more vulnerable to being wind-blown, and potentially settling and clogging pores upon re-deposition. This essentially results in the special case of a layered soil profile termed ‘capped’ or ‘sealed’. A sealed layer may occur with aggregate breakdown, lack of grain cohesion, hard setting or the destructive action of raindrop impacts on the soil. The seal forms a layer that impedes water flow and hence reduces infiltration (Römkens et al., 1986a, 1986b). Conversely, recent research has shown that slowly permeable layers resulting in perched water tables may be cracked and breached under drought conditions, resulting in dewatering (Robinson et al., 2016). This mechanism was proposed to have induced an alternative soil moisture state where infiltration was increased to such an extent that the soils didn’t fully rewet in the winter.

Soil mineralogy and grains: If a soil has a high clay content there is potential for swelling following rapid wetting. As described previously (Section 3.1.1), drought may open up macropores through cracking, but rapid rewetting may close them again reducing porosity and infiltration (Giraldez and Sposito, 1985). The thermal expansion of grains may also alter the pore structure as previously mentioned.

Soil solution: Capillarity and the sorption of water into soils depends on the adhesion between water and surface and is described by the contact angle between the water and surface. The electrolyte concentration, especially in the presence of sodium can exert a strong influence on the contact angle and hence the capillarity. (Assouline and Narkis, 2011). Following drought, the soil solution may be much more concentrated, moreover, low electrolyte concentration rainfall can have a devastating effect on soil structure in saline soils, casing structural collapse. There are few saline soils in Britain, but areas around the coast may be more susceptible.

Wetting front instability: This can occur for a variety of reasons which may be structural, or physico-chemical and impact the infiltration process (Raats, 1973; Philip, 1975; Jury et al., 2003).

Abiotic soil water repellency: Capillarity exerts sorption due to the adhesion between water and soil surfaces. Fully wettable surfaces have a contact angle of ~0o; as the contact angle increases so the soil becomes less wettable. Contact angles of 90o and above are characteristic of water repellency. At this point the soil surface acts in a similar manner to a gore-tex coat, where water beads up instead of being sorbed. Infiltration on hydrophobic soils is complex, such that hydrophobicity may reduce, or conversely increase, infiltration under different circumstances, as previously discussed. Much of the interest in water repellency focused on its occurrence after wildfire, and the coating of mineral grains by melted hydrophobic organic compounds. In general, infiltration decreases after wildfire (Doerr et al., 2009b), likely due to a combination of macropores in the soil being blocked by soot and the prevalence of hydrophobic soil surfaces. The biological sources of water repellency can lead to increased infiltration in the presence of macropores (Lebron et al., 2007), or reduced infiltration in their absence.

Most of the mechanisms considered above lead to a reduction in infiltration following dry conditions, however, there are some notable exceptions which to date have limited evidence including the transition to an alternative soil moisture state (Robinson et al., 2016) and enhanced infiltration due to soil water repellency (Lebron et al., 2007).

4.1.2. Biotic factors influencing infiltration

Biological factors also have the potential to impact and alter infiltration rates into soils. Robinson et al., (2019) proposed four primary pathways through which the biology alters infiltration rates, these are:

Inputs of organic matter changing bulk density, porosity and/or pore size distribution (Franzluebbers, 2002; Jarvis et al., 2017; Rawls, Nemes, & Pachepsky, 2004; Yang et al., 2014; Robinson et al., 2021)

Rooting structure and decreases in porosity through compression induced by new root growth, or macropore generation when roots decay (Bodner, Leitner, & Kaul, 2014; Fischer et al., 2015; Koestel & Schlüter, 2019)

Biopore characteristics and abundance resulting from the activity of macrofauna, the “ecosystem engineers” (Berry, 2018; Spurgeon et al., 2013; Smettem, 1992)

Microbial activity, especially in the rhizosphere, which impacts biofilms and hydrophobicity (Hallett, 2008).

4.2. The importance of soil organic matter and rooting to infiltration

Plants provide the organic material that breaks down into SOM. Drought reduces the plant's ability to produce organic matter, leading to a reduction in SOM levels over time. Persistent drought and reduced long-term plant productivity is likely to have the largest impacts. Changes in organic matter are important because there is a strong relationship between soil porosity and bulk density, or its reciprocal porosity (Robinson et al., 2021). The mechanical behaviour of plant roots is important as they move through and explore the soil volume in search of water and nutrients. As they do they break up the soil creating pores through root holes and fractures. Meta-analyses indicate the important contribution plants, particularly woody vegetation, makes to infiltration (Thompson, et al., 2010; Basche and DeLonge, 2019, Robinson et al., 2022). Robinson et al (2022) comparing infiltration under different land uses on the same soil type found that infiltration rates were 2.2 x higher under woodland compared to cropland. Grasslands were more nuanced, with higher infiltration rates than cropland in general; but likely dependent on levels of compaction due to grazing. Native grasses appeared to have similar infiltration rates to woodland with rates dropping as grassland soils become more highly managed with grazing pressure increasing. Temporal studies with the impacts of drought on such systems are less common and represent an important knowledge gap. General shrinkage around plant roots may increase porosity for infiltration with drought, but conversely there might be less root exploration, especially through fine roots.

4.3. Linking soil, geology and landcover to catchment response to re-wetting

In the preceding sections the response of soils to drying and rewetting have been explored, particularly in response to the infiltration of precipitation. Thus, when examining catchment response, it is the spatial distribution of soil types and their physical, chemical, and biological interactions, along with their underlying geology which will combine with land cover to determine the underlying hydrological response to precipitation. In this section we consider the evidence of how the spatial distribution of soil types may influence catchment response to drought and rewetting and combine to define infiltration and run-off behaviour.

In relation to catchment moisture transport, the combined behaviour of different soil types, along with their parent materials and the geology of catchments is ultimately expressed in the hydrographs of streams draining individual catchments. Typical end member behaviour catchments can be those which are primarily (i) base flow driven or (ii) demonstrate flashy response due to rapid overland runoff components. Catchments with high baseflow are often those with high proportions of limestone/chalk or sandstone geology and may typically act as groundwater aquifers. High runoff catchments are typically those where infiltration is slow due to a high proportion of clay soils, leading to greater overland flow. The importance of the spatial distribution of different soil types within catchments, is due to each acting as fundamental components in the hydrological and groundwater cycles. Thus, soils are often considered as hydro-pedological units

(Terzlaff et al. 2014). Within a typical catchment, some of these will act as recharge zones, discharge areas and areas of important dynamic storage change (Soulsby et al. 2015; Blumstock et al. 2016). Geris et al. (2015) examined the hydro-pedological controls on catchment water storage during the very dry and warm summer of 2014 in Scotland. They found that storage changes in histosols were small (< 40 mm) whilst moorland (~100 mm) and forest (~200 mm) podzols showed much greater water storage losses. However, storage in soils were quickly replenished once the drought had terminated with forest soils taking the longest time (3-4 months). Their results also suggested that for long periods during the summer, large areas of catchment soils were disconnected from the river network, with run-off generated mainly from the histosols.

These units are often used in groundwater and hydrological models where the spatial variability of soils within catchments is a key variable. In addition, soil moisture classification schemes such as HOST (Boorman et al. 1995) are used to describe water movement and are particularly useful within modelling approaches (Maŕechal & Holman, 2005). In one study in Scotland, Soulsby et al (2021) examined the soil moisture response and impacts on groundwater and stream flow in relation to the 2018 summer drought. This was a drought which followed two previous dry winters that had failed to replenish the soil moisture and groundwater stores. Whilst summer precipitation was only 63% of the long-term average in 2018, they found soil moisture storage was only half of the summer average, with groundwater levels being ~50 cm lower. However, recovery of soil moisture and groundwater stores returned rapidly to the to normal winter levels during the autumn and winter of 2018.

Whilst the soil parent material and geology of a catchment will influence the abiotic infiltration within the catchment, biological processes, particularly through catchment landcover will further influence the catchment hydrograph. Under normal weather conditions, landcover particularly the area of forest/wood, grassland and arable agriculture along with urban area (soil sealing) are key in determining the catchment storm hydrographs. Work on a small catchment in Germany, demonstrated that during a drought period (2018-2020), soil moisture recharge was much lower and occurred during a shorter time period in winter under forests compared to grassland (Kleine et al. 2020; Kleine et al. 2021). Soulsby et al. (2021) also suggest that drought induced soil and groundwater storage deficits may be enhanced by land-use policies such as reforestation. Whilst trees may reduce short-wave radiation from reaching the ground, they also increase so called ‘green water fluxes’ such as interception losses and transpiration, thereby reducing soil moisture storage.

A final feature which is being increasingly investigated concerns the extent that shallow groundwater can buffer the soil moisture during drought (Zipper et al. 2015). Depending on soil texture and structure there is potential for shallow groundwater replenishment of soil moisture based on the upward pull of moisture through capillary action. This can help sustain vegetation growth. The evidence for this is demonstrated through experiments that demonstrate 'actual evapotranspiration' exceeding 'potential evapotranspiration'. This has been undertaken examining the lowering of the water table in dry ecosystems (Healy & Cook, 2002), modelling (Diouf et al. 2020) and lysimeter studies (Anapalli et al. 2016). Ultimately, for silt or sand-based soils, this upward pull of water through capilliarility will act as a soil moisture buffer, possibly reducing the impact of drought on soils.

4.4. Influence of soil sealing and urbanisation on catchment drought response

Whilst not a direct impact on soils, urbanisation and the sealing of soils have a major impact on the catchment hydrograph. However, the impacts of drought on soils above urban areas in catchments will combine with the rapid transport of water caused by surface sealing. A common image in UK summers is of urban flooding after heavy convective rainfall.

4.5. Influence of soil management and agricultural practice on soil response to drought and infiltration

The UK landscapes and its soils are highly managed. Thus, a major factor as to how soils and catchments will respond to drought and rewetting will result from potentially long-term land management practices. In recent years, one focus of soil research has been examining its resilience to perturbations, particularly those of cultivation and climate. Corstanje et al. (2015) defined soil resilience as resistance (degree of change) coupled with recovery (rate and extent of subsequent recovery) from a disturbance. Examining factors which contributed to physical (void ratio) and biological (respiration) function revealed soil taxonomic class, parent material and soil texture were dominant in producing resilience, whilst SOM and land use were ranked lower in the significant factors for 38 soil types from England and Wales. However, Griffiths et al. (2015) cited bulk density, water retention characteristics, soil C and bacterial community as key properties in soil resilience and Gregory et al. (2007) again cited SOM being of great importance, but also suggested that grassland soils with higher SOM may have greater resilience than arable soils. Thus, soils that are well managed, particularly for SOC are potentially more resilient to drought due to the influence of SOC on properties such as water retention and aggregate stability. In this respect, SOC may delay the impact of drought on soil function, but if the drought is prolonged the resilience may continue to fade. Droste et al. (2020) suggested that soil C insured crop production against increasing adverse weather due to climate change.

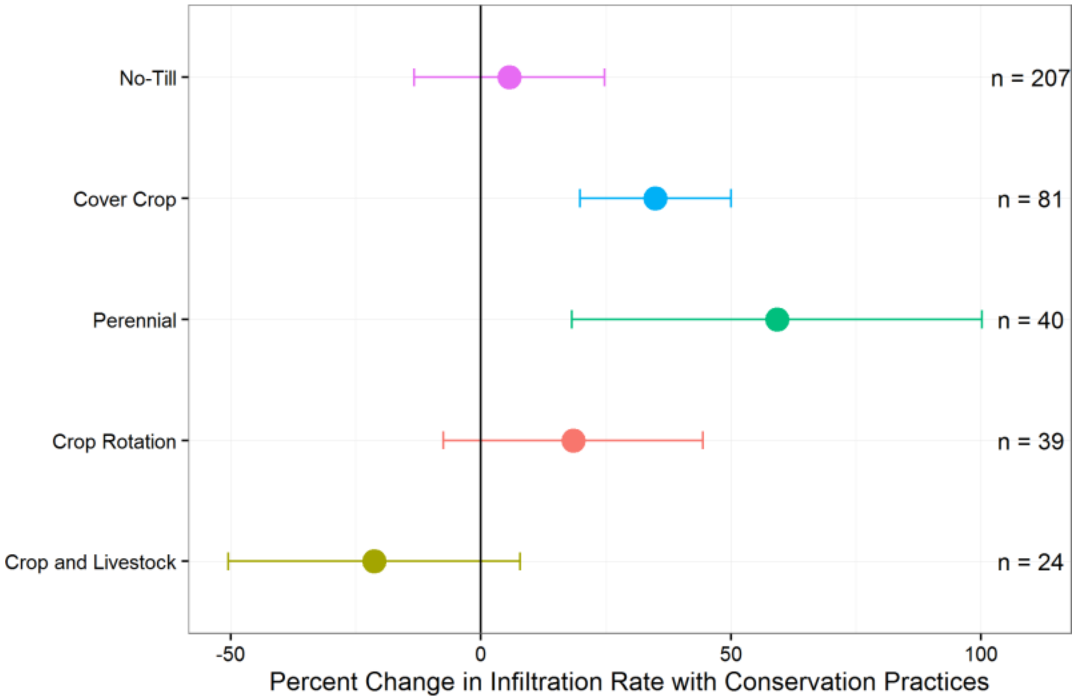


Figure 8: Impact of different agricultural management practices on infiltration rates of cropland soils (Reproduced from Basche and DeLonge (2019) under the CC BY 4.0 licence).

Bearing this in mind, the land cover and the management of agricultural soils prior to drought will have an important impact on topsoil infiltration rates. This was demonstrated in a global meta-analysis by Basche and DeLonge, (2019), who examined a range of soil conservation practices on infiltration rates. Results presented in Figure 8 show their results with perennials and cover crops leading to the largest increases in infiltration. They also showed that the introduction of livestock tended to lead to lower values of infiltration, most likely due to compaction. A wide variety of management practices are considered to aid infiltration in soils. These include (i) the addition of organic matter to soils, (ii) controlled traffic systems to reduce soil disturbance and reducing equipment use when the soils are wet. (iii) limiting the number of times equipment is used on a field, (iv) subsoiling to break up compacted layers, (v) switching cultivation practice to min-till or zero-till, (vi) applying solid manure or other organic material, (vii) using rotations that include high-residue crops, such as corn (maize) and small grain, and perennial crops, such as grass and alfalfa, (viii) using cover crops and (ix) farm on the contour (NRCS, 2011, September 19). However, the extent to which activities such as green manuring affect infiltration is not entirely understood. For example, the practice might lead to the development of water repellency in certain crops which may be exacerbated by drought. Potato fields in Suffolk were found to be strongly water repellent (Robinson, 1999), which anecdotally occurred following the ban on stubble burning. However, on certain soils, under certain green manures, conditions may be conducive to the generation of water repellency, but the extent remains to be determined.

4.6. Within drought management practices that influence infiltration

A major concern relates to high intensity rainfall falling on hard or water repellent soils at the end of 'agricultural drought' causing runoff and erosion. In practice it is likely that agricultural practices, especially in arable systems, may reduce these effects. In many arable systems post-harvest management practice may reduce the area of soil exposed to runoff and erosion. Firstly, the chopping of straw post combine harvester will leave a mulch that will reduce raindrop impact intensity and secondly, post-harvest tillage to help germinate weed seeds before autumn planting is often carried out. This cultivation is likely to increase soil surface roughness thereby decreasing overland flow. Govers et al. (2000) demonstrated that soil surface roughness affects depression storage, water infiltration, overland flow velocity and overland flow organisation, and cultivations are likely to increase all these factors, thereby decreasing the rate of water movement overland and increasing infiltration, potentially decreasing flood and erosion. However, post-harvest cultivations may be considered less attractive until rain has fallen due to the hardness of soil increasing machinery wear and increasing fuel use. These practices on soils vulnerable to erosion or catchments where flooding may occur could be considered in drought years.

4.7. Enhanced run-off and erosion

Two of the key concerns to society of to how soils respond to the end of drought are (i) runoff leading to flooding and (ii) runoff causing soil erosion and the loss of the soil resource. These events were mentioned in the media as potentially occurring at the termination of the 2022 drought, particularly if intense precipitation fell onto hard soils and precipitation > infiltration. Beyond individual soil types susceptibility to drought, vegetation coverage is key to both these processes as it is primarily responsible for (i) reducing the impact of rainfall intensity on processes such as splash erosion and the destruction of soil aggregates, (ii) providing a means of reducing moisture reaching the soil surface by acting as a surface for evaporation and providing a physical barrier to encourage slower overland water movement and (iii) providing a root system that holds the soil together.

Due to the great diversity of soils in the UK, erosion and run-off responses will be spatially different. Various published maps (e.g. Evans, 1990) show those soils most at risk of erosion, these being light sandy and silt based soils. These lighter soils are also likely to be more susceptible to low moisture levels that may have a greater negative impact on crop yield. In general, certain areas are known for their susceptibility to water erosion under current land use (Boardman, 2013) and these include:

- The red sandstone soils of the East (Nottinghamshire) and West Midlands (Shropshire) and south Devon

- Chalklands of southern England because of large areas under winter cereals, large fields steep slopes and compaction along tramlines
- Lower Greensand soils of Bedfordshire, West Sussex and the Isle of Wight. These are often cropped with potatoes, maize, vegetables and cereals
- Silty and fine sandy soils in Somerset around Yeovil (Bridport Sands).

In areas of the world where drought is more common and greater long-term datasets exist, the Universal Soil Loss Equation that combines information on soil texture, slope length and angle, vegetation and precipitation have been combined with vegetation health indexes to assess the potential of drought on erosion (Masroor et al. 2022). Results suggested a positive relationship between drought indices and soil erosion. In the UK, Fullen (1992) demonstrated the importance of vegetation cover in decreasing erosion in plot experiments on a sandy soil in Shropshire. Using a variety of slope angles and lengths, erosion reached 67.4 t ha-1 on bare soils, whilst being negligible on grassed soils with similar slope characteristics. A key concern regarding the end of agricultural drought is that it will occur with a high intensity rainfall event. Fullen & Harrison Reed (1986) demonstrated the impact of rainfall intensity on bare sandy soils in Shropshire. On raindrop compacted (capped) soils, erosion rates were found to be up to 42.7 t ha-1 during individual storms and increased markedly with slope angle. Where slope exceeded 13°, erosion was largely in the form of rill erosion. Whereas prolonged duration, low intensity events were found to cause relatively small amounts of erosion, it was found that high intensity events where precipitation was > 10 mm hr-1 were the drivers for major erosion. The role of extreme drought and high intensity rainfall can be seen in the landscapes of places such as the Badlands in southern Spain (e.g. Ballesteros Cánovas et al., 2017). In these environments, prolonged drought has destroyed vegetation leading to landscapes dominated by deep gully erosion. However, UK agricultural droughts are unlikely to lead to this degree of vegetation destruction. Grassland will generally brown as demonstrated in satellite images often released to the media (Figure 1), but will leave root systems in place, offering some maintenance of the stability of the soil surface. Soils on which arable crops are grown prior to harvest will be protected by vegetation coverage and after harvest will likely maintain root systems until cultivation. Maize and root crops where bare soil exists in between rows and the risks of erosion occurring during normal growing seasons are appreciated (Vogel et al. 2016).

There are numerous studies from around the world linking wind erosion in soils and drought. In many regions of the world this may be combined with over grazing and cropping leaving soil bare (e.g. Kairis et al. 2015). In England there was a history of wind erosion in the past, particularly with the removal of hedgerows in the post war years, but conservation measures have reduced these occurrences. In England, soils susceptible to wind erosion are found in East Yorkshire (Radley & Simms, 1967), Shropshire (Fullen, 1985), Lincolnshire (Robinson, 1968), the East Midlands (Wilkinsion et al. (1968), and the East Anglian Fens (Pollard & Millar, (1968). Chappell & Warren (2003) undertook a quantitative study of wind erosion in England. They examined wind erosion flux in eastern England using 137Cs and found net soil loss amounted to ~0.6 t ha a-1 with a range of -32.6 to +37.5 t ha a-1.

Whilst silty and sandy soils may be at greatest risk of erosion, clay soils are likely to behave differently and with varying results. They are more likely to initially have higher soil moisture retention abilities, but upon severe drying may produce hard surfaces, enhancing run-off. Smectitic clay soils that are prone to cracking in dry periods may demonstrate rapid recharge of soil moisture after storm events, reducing the pressure on runoff and on potential soil erosion. However, because of poor drainage many clay soils, have had some form of under-field drainage installed. Combined with cracking these drainage systems can act as a very fast flow conduit to streams and rivers, thus increasing the flashiness of the storm hydrograph and potentially contributing to increased river flow. However, the greatest contribution to flooding after droughts is probably the degree of soil sealing within a catchment. Records of flooding after drought are usually more often associated with urban flooding as a result of intense rainfall. However, rapid runoff via the surface or via drainage systems will likely increase the pressure on flooding.

5. Soil recovery after drought

5.1. Drought termination

The nature of the rainfall will often determine how soils respond. Intense, convectional storms (more typical of drought terminations during the summer half-year) may deliver rainfall at an intensity that leads to infiltration < excess thus promoting run-off and localised flooding, while steady frontal rain (more typical of drought terminations during the winter half-year) may slowly rewet the soil. In addition, after significant perturbation of the soil system is it likely that physical, biological and chemical recovery from drought will operate over a variety of timescales, and some parts of the system may reflect an altered state.

5.2. Rewetting soils

In the ideal case that rainfall infiltrates into soils the soils simply rewet. As indicated in Section 3, infiltration is not a straightforward process and multiple factors may alter behaviour from the expected norm. The speed of rewetting will impact soil processes and infiltration, especially in regard to the entrapment of air and time to ponding. For soils that may have altered structurally during drought, for example through shrink-swell processes, the cracking is unlikely to change back, but remain as structural weaknesses which will promote future cracking. It will also impact biological processes, unprotected cells subject to rapid rewetting may simply burst without time to adjust. This is why many soil organisms produce biofilms which can provide a protective environment.

Kaisermann et al. (2017) investigated the legacy effects of drought on bacterial and fungal communities. They describe how drought can have a lasting impact on below ground communities with consequences on plant-soil feedbacks and plant-plant interactions which may change soil functioning and plant community composition. However, these studies are in their infancy but demonstrate the more complex legacies of drought. These impacts are further complicated by the vast array of soil types and the specific climatic, environmental factors and management practices they are exposed to.

5.3. Hydraulic contrast in layered soils

Experimentation and long-term catchment observations (Hudson, 1988; Robinson et al., 2016) provide evidence that under certain circumstances some soils do not simply rewet. Robinson et al. (2016) observed a phase change in soil moisture that may also constitute a shift to an alternative stable soil moisture state. This observation was made on peat topped soils over a slowly permeable mineral layer susceptible to cracking with the onset of drought, underlain by a permeable fractured bedrock. The mechanism proposed that the slowly permeable layer leads to an annual perched water table, enabling the formation of a peaty top-soil, commonly found in histosols, podzols and leptosols (WRB classification). If the mineral layer was thin enough, drought could crack the mineral layer allowing the perched water to drain. As a result, if the cracks do not seal in the winter, build-up of the perched water is prevented, and the soil settles to a new soil moisture state. This mechanism is feasible in any layered soil system as discussed below.

Figure 9 shows a snapshot of modelled soil moisture distribution for a hydraulic contrast layered soil (A) both during a dry (drought) summer period with 0.1 mm of rain in the previous 2 weeks, and a wet period in the winter where the soil is saturated after 8.3 mm of rain fell in the previous 2 weeks. Two scenario’s are depicted, B) with changing B horizon thickness (L2) but constant hydraulic conductivity (K2) and C) with a constant B horizon thickness (L2) but changing hydraulic conductivity (K2). The scenarios describe a layered soil (peaty layer, over a slowly permeable mineral layer, over a freely draining layer). Figure 9 D, scenario B) illustrates the impact of water being impeded by the B horizon (L2), which increases as the layer gets thicker. As the layer becomes thicker the soil moisture levels are higher in the A horizon (L1) during wet periods; there’s little impact on the dry soil. The higher moisture content in the upper horizon (L1) would be sustained, as long as the hydraulic conductivity of L2 doesn’t suddenly increase, for example through macropore development via cracking or plant roots. Scenario C) is where the layer thickness is fixed, but the B horizon (L2) hydraulic conductivity alters. Again, the decreasing hydraulic conductivity of L2 causes a hydraulic bottle neck which impedes water flow. The water content in L1 increases as L2 decreases unless there is a dramatic change in the hydraulic conductivity of L2. As with scenario B this is most likely to occur due to the generation of a macropore network developing, for example due to drought and cracking through soil shrinkage (Beven and Germann, 1982; Messing and Jarvis, 1990; Lin et al., 1998). Messing and Jarvis (1990) presented a power law function for their data showing that an order of magnitude increase in macro-porosity led to an increase in hydraulic conductivity of 3 orders of magnitude (~0.2 – 200 cm day-1) in their soils. The examples here indicate that about one order of magnitude difference between layer hydraulic conductivities is required to begin marked changes in the soil moisture frequency distribution in the field. As the hydraulic conductivity contrast increases between layers, the A horizon (L1) becomes much wetter. Any breach in the impeding layer with hydraulic contrasts >1 order of magnitude would be expected to cause a step change in the A horizon soil moisture storage due to the layer draining, resulting in an alternative stable soil moisture state in this soil layer.

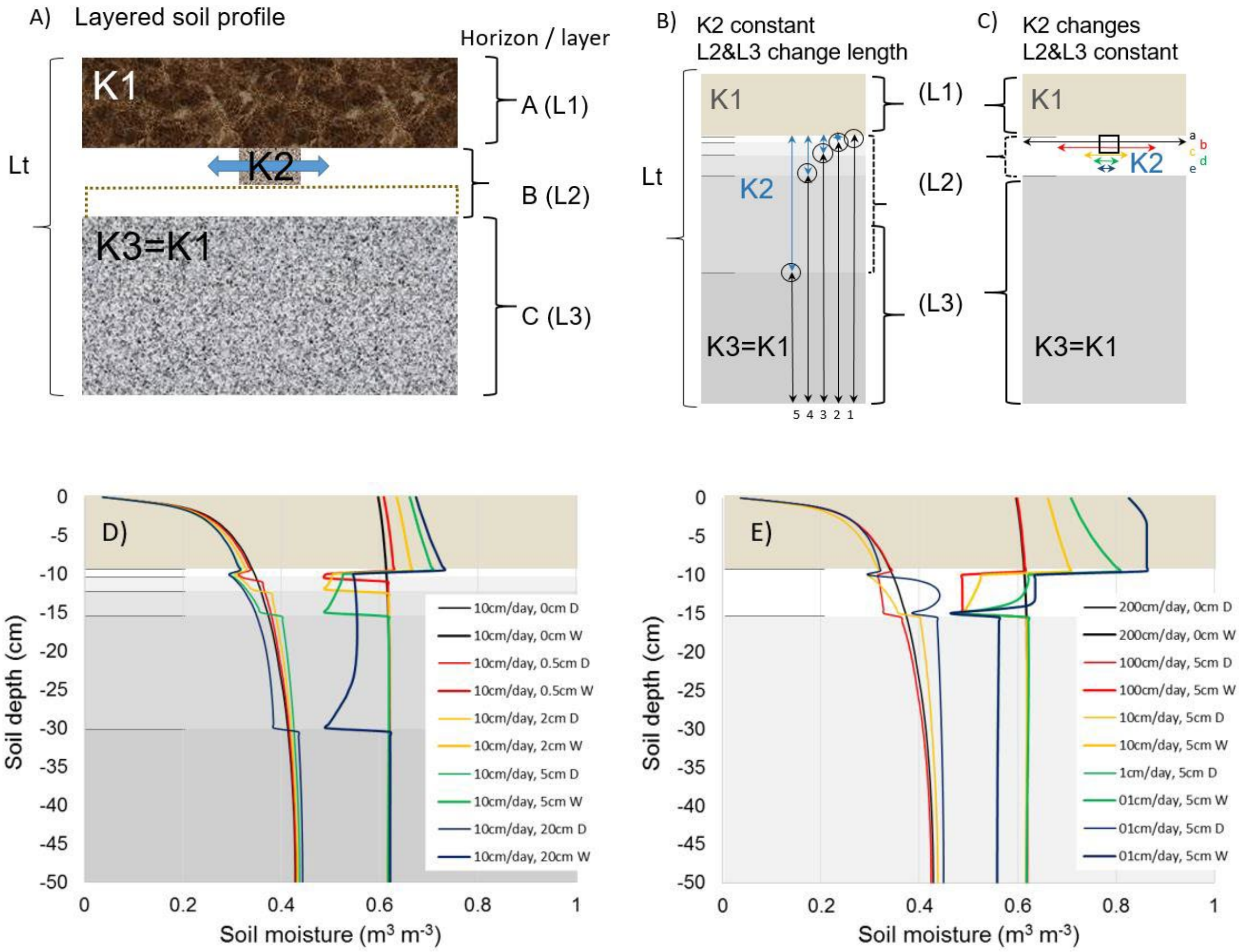


Figure 9: A) hydraulic-contrast layered soil with an A and C horizon with a high hydraulic conductivity (K1, 200cm/day), the B horizon (L2) has a low hydraulic conductivity (K2). B) The length of the B and C horizon can be altered while K2 is kept constant (10cm/day). C) The length of the B horizon is kept constant (5cm) but K2 is altered becoming less impermeable (K2 a-e, 200, 100, 10, 1 and 0.5 cm/day). The effect this has on the soil moisture distribution is shown in D for case B and E for case C. The lines on the left are for a dry period and the lines on

the right for a wet period. Produced by David Robinson (UKCEH) for the purposes of the project

The major unknown is the extent to which this phenomenon occurs. Long-term soil moisture measurements at the Plynlimon observatory over a number of drought periods indicate that this happens and is dependent on the soil type and the vegetation (Hudson, 1988); the role of the vegetation may be important with woody plants putting roots deeper into the soil during drought to find water and causing breaches to a previously impermeable soil layer. Perched water tables in soils are not uncommon and impermeable layers can form through natural processes or management practices such as the formation of plough pans or compacted layers.

6. Key questions and unknowns

This essay has demonstrated that considerable information is available regarding the effects of drying on soils. However, this information is often gained from experiments based on how wetting and drying cycles impact soil properties or from climate manipulation experiments. Evidently, due to the relatively low frequency of droughts in the UK, much of this information does not result from studies specifically undertaken to examine responses in soils to more severe periods of drought. Whilst a couple of long-term monitoring experiments of soil moisture have run when periods of drought and re-wetting occurred, thus providing evidence of the recovery of soils, these experiments are rare and this means that relatively little is known about how soils respond and recover from extended drought periods. This contrasts to groundwater and surface water responses to drought which are captured across the country because of the long-term monitoring programs required to understand flooding and the planning of ground water resources. A fundamental requirement to understand the impact and recovery from drought in soil systems is that data is required prior, during and after drought. As UK droughts are unpredictable it is therefore harder to establish meaningful experiments. Thus, a number of questions remain to be answered with specific regard to extended periods of heat and dry soil, for both individual soil types and how these may contribute to catchment responses as drought terminates. These include:

1. Increasingly, minimum till and zero till practices are used on arable land, as these practices have the potential for increasing SOM content and decreasing fuel use. These practices may change soil properties linked to infiltration such as consolidation and roughness (Bauer et al. 2015). It has also been found that increases in preferential flow may occur under min/zero till systems through the development of macro-faunal pathways creating greater connectivity for drainage (Galdos et al. 2019). Key questions are:

a. Do modern tillage techniques such as min/zero till increase runoff compared to traditional soil management after dry periods?

b. Intensively cropped grassland, especially fields used for silage production and where controlled traffic management isn't used can result in soil compaction which will decrease infiltration rates and increase overland flow. How important is this to runoff and infiltration within catchment response?

2. Generally, little is known about soil water repellency (SWR) across the UK soils. Therefore, greater information is required regarding:

   a. The spatial extent of SWR, where is it greatest and how much and at what rate does it change during extended periods of drought?

   b. The extent to which SWR impacts infiltration following drought and whether this has an overall effect on the catchment response?

3. Soils which demonstrate hydraulic contrasts within their profiles are of interest as infiltration processes may be fundamentally changed after drought. Thus, questions that need to be addressed include:

   a. How widespread are hydraulic contrast soils and which soils are most susceptible to changing infiltration function through drought; is this a problem or not?

   b. How does drought affect hydraulic contrast soils and soil moisture movement in the soil profile, is it important?

   c. What are the ecological consequences of drought on hydraulic contrast soils?

4. Recent work has started to understand how shallow water tables may buffer soil moisture (e.g. Zipper et al. 2015), in terms of crop production and the potential buffering of soil moisture to root systems during dry periods:

   a. What is the extent, and in which soils may shallow groundwater help buffer soil moisture for crops, and how long may this process endure in periods of drought?

   b. How do shallow water tables recover after drought periods in different soil types and with human interventions such as under field drainage systems

5. With increasing importance being aimed at maintaining and increasing organic C storage in soils, the impacts of drought on C dynamics needs to have greater

understanding. This essay has shown how drought may impact soil aggregates which often act to protect SOC. Key questions would include:

   a. How vulnerable are soil C stores to drought, especially in organo-mineral soils with limited water reserves?

   b. What is the impact of rewetting on the magnitude of GHG production in different soil types after drought termination?

   c. What is the impact of drought on surface water quality with respect to DOC loss after drought termination?

6. Soil bacterial and fungal communities, and soil fauna have been shown to be susceptible to drought. This often impacts soil C and nutrient recycling but also their role in in maintaining soil structure and hence infiltration. Thus, further work is required to understand:

   a. Whether tipping points exist where soil biological functions do not recover after drought or drying-rewetting? What is the impact on soil function of a new soil state / function?

   b. Do soil biological components recover ecosystem function after a combination of drought-rewetting-disturbance (e.g. fire/flood/pests) and what are the reasons for this resilience to another environmental stressor?

   c. Will an increased frequency of ‘agricultural droughts’ expose subsoils to drought more frequently? If so, we need to understand the biological adaptation of subsoils to droughts, and the potential consequences for SOC and nutrient mineralisation. In addition, what is the potential for nutrient ‘priming’ SOC and will this ultimately impact soil structure.

7. With ongoing climate change, it is not just drought which can provide severe perturbations to the soil system. Increasingly, extreme weather occurrences happen in quicker succession. This means that soils may be subject to further extreme weather events, before fully recovering from previous perturbations, experiencing multiple stressors in quick succession. Key questions are:

   a. What impact will multiple stressors (e.g. flood, drought, temperature extremes both high and low) have on the function of soils?

   b. Where will multiple stresses most likely occur in the UK?

   c. How will more frequent floods or droughts impact soil structure and function?

d. Which soils are most resilient to drought and multiple stressors, why, and how may we increase soil resilience?

8. An area of increasing interest is the role and feedbacks between soils, vegetation and atmosphere and how these may influence the intensity of droughts and heatwaves. This research is in its infancy, but a key question would be:

    a. How strong is the link between soil moisture, drought and the persistence and magnitude of heatwaves across UK?

9. Soils are an integral component of aquifer dynamics, being the initial receptor for precipitation. As discussed in this essay changes in the physical structure of soils may occur during droughts and, particularly in the case of shrink-swell processes soils, these can extend to ~3 m. Thus, a key question would be:

    a. What role does drought have on aquifer recharge, and is recharge slower or faster after drought and what role do soils and land-cover play?

10. The recovery of soil functions with the termination of drought has been studied across ecosystems and for different components of the soil system (see above e.g. bacteria, fungi, C, nutrients). Different intensities of drought have been experimentally imposed to ecosystems and soil types for decadal periods (e.g. Robinson et al. 2016, Seaton et al. 2022, Sayer et al. 2017). However, these experiments are not commonly used to test the effect of additional disturbances (e.g. wildfire) on soil function recovery. From geographical regions where wildfires are common (e.g. southern Europe), the combination of drought x fire showed effects on soil chemistry and soil biology (Hinojosa et al. 2019). In addition, forest windthrows (or felling of mature forests for timber production) and pest outbreaks are relevant examples where recovery of soils may fail when paired with (severe) droughts and there is a knowledge gap that exists around these issues.

11. Research on microbial community responses to peatland re-wetting (after drainage) suggests some recovery of soil function after long-term drought events. Emsens et al. (2020) observed that microbial communities in 13 fens recovered with rewetting, but only if the SOM content was above 72% of the initial content. This finding suggests that soil biological functions can recover if the physical and chemical environment is still favourable. Further, Gao et al. (2020) performed a meta-analysis on the effects of wetting and drying cycles on soil C and phosphorus. The data showed that the soil microbial communities recovered from drought after re-wetting across soil and land-use types. Drying and rewetting was found to have different effects on the magnitude of components in the soil N and P cycle, potentially slowly changing the chemical environment for soil biota. Once a better understanding of future drying and rewetting cycles, their frequency and magnitude is gained, research should be targeted to explore how soil biological components respond to these new pressures given a

change in the soil physical and chemical environment. This knowledge is required across a range of soil types vs. land uses.

12. Knowledge is required on how different drought-exposed soil ecosystems (and on different soil types) recover after a major perturbation such as fire or flooding. Will the previous ecosystems and their functions return to their pre-disturbance state, or will a new ecosystem establish? What consequences will a change in ecosystem structure and functioning have in terms of soil resources (C and nutrients) in an agricultural context as well as from a financial perspective?

7. How to improve understanding

In Section 6 it was noted that one of the main issues with assessing the impact of drought on soils was their infrequency and the difficulty in setting up experiments when there is little knowledge when droughts may occur. In this section we offer routes through which greater understanding could be gained.

How does drought impact the behaviour of HOST soil drainage classes?

A relatively straightforward desk study approach would be to revisit the HOST classes and assess how they may respond to drought with reference to infiltration processes. Classes may need to be broken down further, dependent on soil parent material, considering clay mineralogy and land cover, to further refine the interactions described in this essay such as SWR and shrink swell.

Monitoring and long-term infrastructure

The impacts of drought could be further understood using long-term experiments. These provide the before and after allowing the magnitude and recovery of drought impacts to be analysed. Recently, work has been undertaken for the Welsh Governments using high resolution aerial photography to assess soil erosion and disturbance (Tye et al. 2022). Using high temporal resolution aerial photography, such as provided by Sentinel-2 or Planet Lab satellites, information could be obtained on erosion deposition fans and the area of bare soil that droughts may create. This analysis could be followed up walk over surveys to assess the extent of processes such as rill erosion after drought termination. The UKCEH – COSMOS network soil moisture network is able to understand the 0-15 cm soil moisture response, but this does not cover deeper soil where soil moisture storage occurs. As part of the NERC-BBSRC funded ASSIST program data, data on soil moisture and shallow groundwater interactions on a clay soil in southern England has been gathered for the 2022 drought. In addition UKRI has invested £38 million in Flood Defence Research Infrastructure (FDRI) and this should provide greater information on catchment response to droughts and floods.

New technologies

Satellite technologies now exist such as satellite interferometry (e.g. DInSAR – differential interferometric synthetic aperture radar), using the Sentinel 2 network, that are being assessed to monitor land movement changes (e.g. peat shrinkage), vegetation changes (vegetation dying and over grazing) and soil moisture. Soulsby et al. (2012) suggest that a greater focus should be made on understanding soil moisture deficits and this could be achieved through the wider use of cosmic ray neutron sensors and hydro-geophysics such as the BGS PRIME 4D electrical resistivity tomography system used for monitoring sub-surface moisture in soils and infrastructure (Holmes et al. 2022).

References

Abbasi, A. O., Salazar, A., Oh, Y., Reinsch, S., del Rosario Uribe, M., Li, J., Rashid, I., and Dukes, J. S. 2020. Reviews and syntheses: Soil responses to manipulated precipitation changes – an assessment of meta-analyses, Biogeosciences, 17, 3859–3873, https://doi.org/10.5194/bg-17-3859-2020

Anapalli, S. S., Ahuja, L. R., Gowda, P. H., Ma, L., Marek, G., Evett, S. R., & Howell, T. A. 2016. Simulation of crop evapotranspiration and crop coefficients with data in weighing lysimeters. Agric. Water Mgmt., 177, 274-283. https://doi.org/10.1016/j.agwat.2016.08.009

Anderegg, W.T.L., Berry, J.A., Smith, D.D., Sperry, J.S., Anderegg, L.D.L, Field, C.B. 2011. The roles of hydraulic and carbon stress in a widespread climate-induced forest die-off. PNAS, 109, 233-237.

Assouline, S., and K. Narkis 2011. Effects of long-term irrigation with treated wastewater on the hydraulic properties of a clayey soil, Water Resour. Res., 47, W08530, doi:10.1029/2011WR010498.

Assouline, S., 2013. Infiltration into soils: Conceptual approaches and solutions. Water Resources Research, 49(4), pp.1755-1772.

Barnard, R.L., Blazewicz, S.J., Firestone, M.K. 2020. Rewetting of soil: Revisiting the origin of soil CO2 emissions, Soil Biology and Biochemistry, 147, 107819,

Basche, A.D. and DeLonge, M.S., 2019. Comparing infiltration rates in soils managed with conventional and alternative farming methods: A meta-analysis. PloS One, 14(9), p.e0215702.

Bauer, T., Strauss, P., Grims, M., Kamptner, E., Mansberger, R., Spiegel, H. 2015. Long-term agricultural management effects on surface roughness and consolidation of soils, Soil and Tillage Research, 151, 28-38,

Bi, W., Zhang, D., Weng., Dong, Z., Wang, F., Wang, W., Lin, W., Yan, D. 2023. Research progress on the effects of droughts and floods on phosphorus in soil-plant ecosystems based on knowledge graph, Hydro Research,

Birch, H. 1958. The effect of soil drying on humus decomposition and nitrogen availability. Plant Soil, 10, 9-31.

Blumstock, M., Tetzlaff, D., Dick, J. J., Nuetzmann, G., and Soulsby, C. (2016) Spatial organization of groundwater dynamics and streamflow response from different hydropedological units in a montane catchment. Hydrol. Process., 30: 3735– 3753

Boardman, J. 2013. Soil Erosion in Britain: Updating the Record. Agriculture, 3, 418-442

Bogati, K., Walczak, M., 2022. The Impact of Drought Stress on Soil Microbial Community, Enzyme Activities and Plants, Agronomy 12, 189.

Bond-Lamberty, B. and Thomson, A., 2010. Temperature-associated increases in the global soil respiration record. Nature, 464(7288), pp.579-582.

Bradley, C., Mosugu, M.E., Gerrard, A.J. 2005. Simulation modelling of variable patterns of water movement through a cracking soil. Soil Use and Management, 21, 386-395.

Berry, E. C. 2018. Earthworms and other fauna in the soil. In J. L. Hatfield & B. A. Stewart (Eds.), Soil biology (pp. 67– 96). Boca Raton, FL: Lewis Publishers, CRC Press.

Beven, K. and Germann, P., 1982. Macropores and water flow in soils. Water resources research, 18(5), pp.1311-1325.

Bodner, G., Leitner, D., & Kaul, H.-P. 2014. Coarse and fine root plants affect pore size distributions differently. Plant and Soil, 380, 133– 151.

Canarini, A., Kiær, L.P. and Dijkstra, F.A., 2017. Soil carbon loss regulated by drought intensity and available substrate: A meta-analysis. Soil Biology and Biochemistry, 112, 90-99.

Canarini, A., Schmidt, H., Fuchslueger, L. et al. 2021. Ecological memory of recurrent drought modifies soil processes via changes in soil microbial community. Nat Commun 12, 5308

Capowiez, Y., Sammartino, S., Michel, E. 2014. Burrow systems of endogeic earthworms: Effects of earthworm abundance and consequences for soil water infiltration. Pedobiologia 57: 303-309.

Capowiez, Y., Bottinelli, N., Sammartino, S., Michel, E., Jouquet, P. 2015. Morphological and functional characterisation of the burrow systems of six earthworm species. Biology and Fertility of Soils 51: 869-877.

Carey, J.C., Tang, J., Templer, P.H., Kroeger, K.D., Crowther, T.W., Burton, A.J., Dukes, J.S., Emmett, B., Frey, S.D., Heskel, M.A. and Jiang, L., 2016. Temperature response of soil respiration largely unaltered with experimental warming. Proceedings of the National Academy of Sciences, 113(48), pp.13797-13802.

Chen, H., Wang, Y., Wu, N., Zhu, D, Li, W., Gao, Y., Qui'An, Z., Peng, C. 2012. Spatiotemporal variations in nitrous oxide emissions from an open fen on the Qinghai-Tibetan plateau. Water, Air Soil Pollution, 223, 6025-6034.

Choat, B., Brodribb, T.J., Brodersen, C.R. et al. 2018.Triggers of tree mortality under drought. Nature 558, 531–539. https://doi.org/10.1038/s41586-018-0240-x

Crowther, T.W., Todd-Brown, K.E., Rowe, C.W., Wieder, W.R., Carey, J.C., Machmuller, M.B., Snoek, B.L., Fang, S., Zhou, G., Allison, S.D. and Blair, J.M., 2016. Quantifying global soil carbon losses in response to warming. Nature, 540(7631), pp.104-108.

Da Silva, E.C., Nogueira, R.J.M.C., da Silva, M.A., de Albuquerque, M.B. 2009. Drought Stress and plant nutrition. Plant Stress 5 (special Issue 1), 32-41. Global Science Books.

Diouf, O.C., Weihermüller, Diedhiou, M., Vereecken, H., Faye, S.C., Faye, S., Sylla, S.N. 2020. Modelling groundwater evapotranspiration in a shallow aquifer in a semi-arid environment. Journal of Hydrology, 587, 124967.

Dobbie, K.E., Bruneau, P.M.C and Towers, W. (eds) 2011. The State of Scotland's Soil. Natural Scotland, www.sepa.org.uk/land/land_publications.aspx

Doerr, S.H., Shakesby, R.A. and Walsh, R., 2000. Soil water repellency: its causes, characteristics and hydro-geomorphological significance. Earth-Science Reviews, 51(1-4), pp.33-65.

Doerr, S.H., Woods, S.W., Martin, D.A. and Casimiro, M., 2009a. 'Natural background' soil water repellency in conifer forests of the north-western USA: Its prediction and relationship to wildfire occurrence. Journal of Hydrology, 371(1-4), pp.12-21.

Doerr, S.H., Shakesby, R.A. and MacDonald, L.H., 2009b. Soil water repellency: a key factor in post-fire erosion. In Fire effects on soils and restoration strategies (pp. 213-240). CRC Press.

Duchicela, J., Sullivan, T.S., Bontti, E., Bever, J.D. 2013. Soil aggregate stability increase is strongly related to fungal community succession along an abandoned agricultural field chronosequence in the Bolivian Altiplano. Journal of Applied Ecology, 50, 1266-1273.

Emsens, WJ., van Diggelen, R., Aggenbach, C.J.S. et al. 2020. Recovery of fen peatland microbiomes and predicted functional profiles after rewetting. ISME J 14, 1701–1712

Epstein, A.K., Pokroy, B., Seminara, A. and Aizenberg, J., 2011. Bacterial biofilm shows persistent resistance to liquid wetting and gas penetration. Proceedings of the National Academy of Sciences, 108(3), pp.995-1000.

Estop-Aragonés, C., Knorr, K.H., Blodau, C. 2012. Controls on in situ oxygen and dissolved inorganic carbon dynamics in peats of a temperate fen. J. Geophysical Research, 117.

Estop-Aragonés, C., Zajac, K., Blodau, C. 2016. Effects of extreme experimental drought and rewetting on CO2 and CH4 exchange in mesocosms of 14 European peatlands with different nitrogen and sulfer deposition. Global Change Biology, 22, 2285-2300.

Eggleton, P., Inward, K., Smith, J., Jones, D.T., Sherlock, E. 2009. A six year study of earthworm (Lumbricidae) populations in pasture woodland in southern England shows their responses to soil temperature and soil moisture. Soil Biol. Biochem. 41, 1857–1865.

Evans, R. 1990. Soils at risk of accelerated erosion in England and Wales. Soil Use Manage. 6, 125– 131.

Fischer, C., Tischer, J., Roscher, C., Eisenhauer, N., Ravenek, J., Gleixner, G., … Hildebrandt, A. 2015. Plant species diversity affects infiltration capacity in an experimental grassland through changes in soil properties. Plant and Soil, 397, 1895–16.

Franzluebbers, A. J. 2002. Water infiltration and soil structure related to organic matter and its stratification with depth. Soil and Tillage Research, 66, 197– 205.

Fuchslueger, L., Bahn, M., Hasibeder, R., Kienzl, S., Fritz, K., Schmitt, M., Watzka, M. and Richter, A. 2016. Drought history affects grassland plant and microbial carbon turnover during and after a subsequent drought event. J Ecol, 104: 1453-1465.

Fullen, M.A., 1985. Wind erosion of arable soils in east Shropshire (England) during spring 1983. Catena, 12(2), 111-120.

Galdos,M.V., Pires, L.F., Cooper, H.V., Calonego, J.C., Rosolem, C.A., Mooney, S.J. 2019. Assessing the long-term effects of zero-tillage on the macroporosity of Brazilian soils using X-ray Computed Tomography. Geoderma,337, 1126-1135,

Gao, D., Bai, E., Li, M., Zhao, C., Yu, K. and Hagedorn, F., 2020. Responses of soil nitrogen and phosphorus cycling to drying and rewetting cycles: A meta-analysis. Soil Biology and Biochemistry, 148, p.107896.

Gao, H. and Shao, M., 2015. Effects of temperature changes on soil hydraulic properties. Soil and Tillage Research, 153, pp.145-154.

Gao, H., Fenicia, F., Savenije, H. H. G. 2023. HESS Opinions: Are soils overrated in hydrology? EGUsphere [preprint], https://doi.org/10.5194/egusphere-2023-125

García-Palacios, P., Crowther, T.W., Dacal, M., Hartley, I.P., Reinsch, S., Rinnan, R., Rousk, J., van den Hoogen, J., Ye, J.S. and Bradford, M.A., 2021. Evidence for large microbial-mediated losses of soil carbon under anthropogenic warming. Nature Reviews Earth & Environment, 2(7), pp.507-517.

Gao, W., Reed, S.C., Munson, S.M., Rui, Y.C., Fan, W., Zheng, Z., Li, L., Che, R., Xue, K., Du, J., 2021. Responses of soil extracellular enzyme activities and bacterial community composition to seasonal stages of drought in a semiarid grassland. Geoderma, 401, 115327.

Geng, S.M., Yan, D.H., Weng, B.S., Zhang, Z.B., Qin, T.L. 2015. Effects of drought stress on agricultural soils. Natural Hazards, 75, 1997-2011.

Geris, J., Tetzlaff, D., and Soulsby, C. 2015. Resistance and resilience to droughts: hydropedological controls on catchment storage and run-off response. Hydrol. Process., 29: 4579– 4593. Doi: 10.1002/hyp.10480.

Giraldez, J. V., and G. Sposito 1985. Infiltration in swelling soils, Water Resour. Res., 21, 33–44.

Gongalasky, K. B. 2021. Soil macrofauna: Study problems and perspectives. Soil Biology & Biochemistry 159: 108281.

Govers, G., Takken, I., Helming, K. 2000. Soil roughness and overland flow. Agronomie, 20, 131-146.

Graber, E.R., Tagger, S. and Wallach, R., 2009. Role of divalent fatty acid salts in soil water repellency. Soil Science Society of America Journal, 73(2), pp.541-549.

Harris, G.L., Catt, J.A. 1999. Overview of the studies on the cracking clay soil at Brimstone Farm, U.K. Soil Use and Management, 15, 233-239.

Hallett, P. D. 2008. A brief overview of the causes, impacts and amelioration of soil water repellency–a review. Soil and Water Research, 3, S21– S29.

Harrison AM, Plim JFM, Harrison M, Jones LD, Culshaw MG. 2012. The relationship between shrink-swell occurrence and climate in south-east England. Proceedings of the Geologists’ Association. Vol. 123, pp. 556-575.

Hatano, R., Nakamoto, H., Sakuma, T., Okajima, H. 1988. Evapotranspiration in cracked clay field soil. Soil Science and Plant Nutrition, 34, 547-555.

Hawkins AB. 2013. Some engineering geological effects of drought: examples from the UK. Bulletin of Engineering Geology and the Environment. Vol. 72, pp. 37-59.

Healy, R. W., & Cook, P. G. 2002. Using groundwater levels to estimate recharge. Hydrogeology Journal, 10(1), 91– 109. https://doi.org/10.1007/s10040-001-0178-0

Hinojosa, M.B., Laudicina, V.A., Parra, A., Albert-Belda, E. and Moreno, J.M., 2019. Drought and its legacy modulate the post-fire recovery of soil functionality and microbial community structure in a Mediterranean shrubland. Global Change Biology, 25(4), 1409-1427.

Holmes, J., Chambers, J., Wilkinson, P., Dashwood, B., Gunn, D., Cimpoiaşu, M., Kirkham, M., Uhlemann, S., Meldrum, P., Kuras, O., Huntley, D., Abbott, S., Sivakumar, V. and Donohue, S. 2022. 4D electrical resistivity tomography for assessing the influence of vegetation and subsurface moisture on railway cutting condition. Engineering Geology 307.

Holmstrup, M. 2001. Sensitivity of life history parameters in the earthworm Aporrectodea caliginosa to small changes in soil water potential. Soil Biology & Biochemistry 33: 1217–1223.

Horton, R. E. 1940. An approach towards a physical interpretation of infiltration capacity, Soil Sci. Soc. Am. Proc., 5, 399–417.

Hudson, J.A., 1988. The contribution of soil moisture storage to the water balances of upland forested and grassland catchments. Hydrological sciences journal, 33(3), pp.289-309.

Hughes, S., Reynold, B., Hudson, J.A., Freeman, C. 1997. Effects of summer drought on peat soil solution chemistry inan acid gully mire. Hydrology and Earth System Science, 1, 661-669.

Jarvis, N.J., Leeds-Harrison, P.B. 1987. Modelling water movement in drained clay soil. II. Application of the model in Evesham series clay soil. Journal of Soil Science, 38, 499-509.

Jarvis, N.J., Leeds-Harrison, P.B. 1990. Field test of a water balance model of cracking clay soils, Journal of Hydrology, 112, 3-4, 203-218.

Jarvis, N., Forkman, J., Koestel, J., Kätterer, T., Larsbo, M., & Taylor, A. 2017. Long-term effects of grass-clover leys on the structure of a silt loam soil in a cold climate. Agriculture, Ecosystems & Environment, 247, 319– 328.

Jia, X., Wu, F., Wu, Q., Heděnec, P., Peng, Y., Wang, Z., Yue, K. 2023. Effects of drying-rewetting cycles on the fluxes of soil greenhouse gases. Heliyon, e12984.

Jones, L., Banks, V., Jefferson, I. 2020. Swelling and shrinking soils. In Giles, D. P. & Griffiths, J. S. (eds) Geological Hazards in the UK: Their Occurrence, Monitoring and Mitigation – Engineering Group Working Party Report. Geological Society, London, Engineering Geology Special Publications, 29, 223–242,

Jouquet, P., Dauber, J., Largerlof, J., Lavelle, P., Lapage, M. 2006. Soil invertebrates as ecosystem engineers: Intended and accidental effects on soil and feedback loops. Applied Soil Ecology 32: 153-164

Jury, W. A., Z. Wang, and A. Tuli 2003. A conceptual model of unstable flow in unsaturated soil during redistribution, Vadose Zone J., 17, 61–67.

Kannenberg, S.A., Phillips, R.P. 2017. Soil microbial communities buffer physiological responses to drought stress in three hardwood species. Oecologia 183, 631–641..

Kairis, O., Karavitis, C., Salvati, L., Kounalaki, A., Kosmas, K. 2015. Exploring the Impact of Overgrazing on Soil Erosion and Land Degradation in a Dry Mediterranean Agro-Forest Landscape (Crete, Greece), Arid Land Research and Management, 29,360-374

Kaisermann, A., de Vries, F.T., Griffiths, R.I., Bardgett, R.D. 2017. Legacy effects of drought on plant-soil feedbacks and plant-plant interactions. New Phytologist, 215, 1413-1424.

Kendon, M., Marsh, T., Parry, S. 2013. The 2010-2012 drought in England and Wales. Weather, 68, 88-95.

Klaus, V.H., Friedritz, L., Hamer, U., Kleinebecker, T. 2020. Drought boosts risk of nitrate leaching from grassland ferilistaion. Science of the Total Environment, 726, 137877.

Kleine, L., Tetzlaff, D., Smith, A., Wang, H., and Soulsby, C. 2020. Using water stable isotopes to understand evaporation, moisture stress, and re-wetting in catchment forest and grassland soils of the summer drought of 2018, Hydrol. Earth Syst. Sci., 24, 3737–3752, https://doi.org/10.5194/hess-24-3737-2020

Kleine L, Tetzlaff D, Smith A, Goldhammer T, Soulsby C. 2021. Using isotopes to understand landscape-scale connectivity in a groundwater-dominated, lowland catchment under drought conditions. Hydrological Processes. 35:e14197

Koestel, J., & Schlüter, S. 2019. Quantification of the structure evolution in a garden soil over the course of two years. Geoderma, 338, 597– 609.

Koura, D., Yadav, A.N. 2022. Bacterial Mitigation of Drought Stress in Plants: Current Perspectives and Future Challenges. Curr Microbiol 79, 248.

Krüger, M., Potthast, K., Michalzik, B., Tischer, A., Küsel, K., Deckner, F.F.K., Herrmann, M. 2021. Drought and rewetting events enhance nitrate leaching and seepage-mediated translocation of mictobes from beech forest soils. Soil Biology and Biochemistry, 154, 108153.

Lebron, I., Madsen, M.D., Chandler, D.G., Robinson, D.A., Wendroth, O. and Belnap, J., 2007. Ecohydrological controls on soil moisture and hydraulic conductivity within a pinyon-juniper woodland. Water Resources Research, 43(8).

Leeds-Harrison, P.B., Shipway, C.J.P., Jarvis, N.J., Youngs, E.G. 1996. The influence of soil macroporosity on water retention, transmission and drainage in a clay soil. Soil Use and Management, 2, 47 - 50

Lin, H.S., Mclnnes, K.J., Wilding, L.P. and Hallmark, C.T., 1998. Macroporosity and initial moisture effects on infiltration rates in vertisols and vertic intergrades. Soil science, 163(1), pp.2-8.

McDaniel, J.P., Barbarick, K.A., Stromberger, M.E., Cranshaw, W. 2013. Survivability of Aporrectodea caliginosa in Response to Drought Stress in a Colorado Soil. Soil Science Society of America Journal 77 (5): 1667.

Maherali, H., Pockman, W. T., & Jackson, R. B. 2004. Adaptive variation in the vulnerability of woody plants to xylem cavitation. Ecology, 85, 2184–2199.

Malik, A. A., & Bouskill, N. J. 2022. Drought impacts on microbial trait distribution and feedback to soil carbon cycling. Functional Ecology, 36, 1442– 1456.

Manzoni, S., Schimel, J. P., & Porporato, A. 2012. Responses of soil microbial communities to water stress: Results from a meta-analysis. Ecology, 93(4), 930–938.

Manzoni, S., Schaeffer, S. M., Katul, G., Porporato, A., & Schimel, J. P. 2014. A theoretical analysis of microbial eco-physiological and diffusion limitations to carbon cycling in drying soils. Soil Biology and Biochemistry, 73, 69–83

Marsh, T.J. 2003. The UK drought of 2003: A hydrological review. Weather, 59(8), 224-230.

Marsh, T., Cole, G., Wilby, R. 2007. Major droughts in England and Wales, 1800-2006. Weather, 62, 87-93.

Messing, I. and Jarvis, N.J., 1990. Seasonal variation in field-saturated hydraulic conductivity in two swelling clay soils in Sweden. Journal of soil science, 41(2), pp.229-237.

Miralles, D.G., Gentine, P., Seneviratne, S.I., Truling, A.J. 2019. Land-atmospheric feedbacks during droughts and heatwaves: State of the science and current challenges. Annals of the New York Academy of Sciences, 1436, 19-35.

Miyazaki, T., 2005. Water flow in soils. CRC Press.

NOAA: https://www.ncei.noaa.gov/access/monitoring/dyk/drought-definition

Oades, J.M. 1984. Soil, organic matter and structural stability: mechanisms and implications for management. Plant and Soil, 76, 319-337.

Oades, J.M., Waters, A.G. 1991. Aggregate hierarchy in soils. Australian Journal of Soil Research, 29, 815-28.

Oades, J.M. 1993. The role of biology in the formation, stabilization and degradation of soil structure. Geoderma, 56, 377-400.

Or, D., Smets, B.F., Wraith, J.M., Dechesne, A., Friedman, S.P. 2007. Physical constraints affecting bacterial habitats and activity in unsaturated porous media – a review. Advances in Water Resources, 30, 1505-1527.

Parry, S., Wilby, R.L., Prudhomme, C. and Wood, P.J., 2016. A systematic assessment of drought termination in the United Kingdom. Hydrology and Earth System Sciences, 20, 4265-4281.

Patel, K.F., Fansler, S.J., Campbell, T.P., Bond-Lamberty, B., Peyton-Smith, A., Roy-Chowdhury, T., McCue, L.A., Varga, T., Bailey, V. 2021. Soil texture and environmental conditions influence the biogeochemical responses of soils to drought and flooding. Nature Communications Earth and Environment, 2, 127.

Philip, J. R. 1975. Stability analysis of infiltration, Soil Sci. Soc. Am. Proc., 39, 1042–1049.

Pollard, E., Millar, A7., 1968. Wind erosion in the East Anglian Fens. Weather, 23(10), 415-417.

Poulson, T.G. 2022. Predicting evaporation from moist, cracked soil, based on near-surface wind speed, crack width and crack distance. European Journal of Soil Science, 13215.

Raats, P. A. C. 1973. Unstable wetting fronts in uniform and non-uniform soils, Soil Sci. Soc. Am. Proc., 36, 681–685.

Radley, J., Simms, C., 1967. Wind Erosion in East Yorkshire. Nature, 216(5110), 20-22.

Rawls, W. J., Nemes, A., & Pachepsky, Y. 2004. Effect of soil organic carbon on soil hydraulic properties. Developments in Soil Science, 30, 95– 114.

Rillig, M.C., Ryo, M., Lehmann, A., Aguilar-Trigueros, C.A., Buchert, S., Wulf, A., Iwasaki, A., Roy, J. and Yang, G., 2019. The role of multiple global change factors in driving soil functions and microbial biodiversity. Science, 366(6467), pp.886-890.

Robinson, D.N., 1968. Soil erosion by wind in Lincolnshire, March 1968. East Midlands Geographer, 4(3), 351-362.

Römkens, M. J. M., R. L. Baumhardt, M. B. Parlange, F. D. Whisler, J.-Y. Parlange, and S. N. Prasad 1986a., Rain-induced surface seals: Their effect on ponding and infiltration, Ann. Geophys., 4, 417–424.

Römkens, M. J. M., R. L. Baumhardt, J.-Y. Parlange, F. D. Whisler, M. B. Parlange, and S. N. Prasad 1986b. Effect of rainfall characteristics on seal hydraulic conductance, in Assessment of Soil Surface Sealing and Crusting, edited by F. Callebaut, D. Gabriels, and M. De Boodt, pp. 228235, Ghent Univ. Press, Belgium.

Robinson, D., 1999. A comparison of soil-water distribution under ridge and bed cultivated potatoes. Agricultural Water Management, 42(2), pp.189-204.

Robinson, D.A., Lebron, I., Ryel, R.J. and Jones, S.B., 2010. Soil water repellency: A method of soil moisture sequestration in pinyon–juniper woodland. Soil Science Society of America Journal, 74(2), pp.624-634.

Robinson, D., Jones, S.B., Lebron, I., Reinsch, S., Domínguez, M.T., Smith, A.R., Jones, D.L., Marshall, M.R. and Emmett, B.A., 2016. Experimental evidence for drought induced alternative stable states of soil moisture. Scientific reports, 6(1), pp.1-6.

Robinson, D.A., Hopmans, J.W., Filipovic, V., van der Ploeg, M., Lebron, I., Jones, S.B., Reinsch, S., Jarvis, N. and Tuller, M., 2019. Global environmental changes impact soil hydraulic functions through biophysical feedbacks. Global change biology, 25(6), pp.1895-1904.

Robinson, D.A., Thomas, A., Reinsch, S., Lebron, I., Feeney, C.J., Maskell, L.C., Wood, C.M., Seaton, F.M., Emmett, B.A. and Cosby, B.J., 2022. Analytical modelling of soil porosity and bulk density across the soil organic matter and land-use continuum. Scientific reports, 12(1), pp.1-13.

Robinson, D.A., Nemes, A., Reinsch, S., Radbourne, A., Bentley, L. and Keith, A.M., 2022. Global meta-analysis of soil hydraulic properties on the same soils with differing land use. Science of The Total Environment, 852, p.158506.

Rubin, R.L., van Groenigen, K.J. & Hungate, B.A. 2017. Plant growth promoting rhizobacteria are more effective under drought: a meta-analysis. Plant Soil 416, 309–323.

Saseendran S. Anapalli, Lajpat R. Ahuja, Prasanna H. Gowda, Liwang Ma, Gary Marek, Steven R. Evett, Terry A. Howell, 2016. Simulation of crop evapotranspiration and crop coefficients with data in weighing lysimeters, Agricultural Water Management, 177, 274-283,

Sayer, E.J., Oliver, A.E., Fridley, J.D., Askew, A.P., Mills, R.T. and Grime, J.P., 2017. Links between soil microbial communities and plant traits in a species-rich grassland under long-term climate change. Ecology and Evolution, 7(3), 855-862.

Seaton, F.M., Jones, D.L., Creer, S., George, P.B., Smart, S.M., Lebron, I., Barrett, G., Emmett, B.A. and Robinson, D.A., 2019. Plant and soil communities are associated with the response of soil water repellency to environmental stress. Science of the total environment, 687, pp.929-938.

Seaton, F.M., Reinsch, S., Goodall, T. et al. 2022. Long-Term Drought and Warming Alter Soil Bacterial and Fungal Communities in an Upland Heathland. Ecosystems 25, 1279–1294.

Sharma, P.P., Gupta, S.C., Rawls, W.J. 1991. Soil detachment by single raindrops of varying kinetic energy. Soil Science Society America Journal, 55, 301-307.

Singh, J., Schädler, M., Demetrio, W., Brown, G. G., Eisenhauer, N. 2020. Climate change effects on earthworms - a review. Soil Organisms 91(3): 113–137.

Six, J., Elliot, E.T., Puastian, K. 1999. Aggregate and soil organic matter dynamics under conventional and no-tillage systems. Soil Science Society America Journal, 63, 1350-1358.

Six, J., Elliot, E.T., Puastian, K. 2000. Soil, structure and soil organic matter: ii. A normalized stability Index and the effect of mineralogy. Soil Science America Journal, 64, 1042-1049.

Six, J., Elliot, E.T., Puastian, K. 2000. Soil macroaggregate turnover and microaggregate formation: a mechanism for C sequestration under no tillage agriculture. Soil Biology and Biochemistry, 32, 2099-2013.

Smettem, K.R., 1992. The relation of earthworms to soil hydraulic properties. Soil Biology and Biochemistry, 24(12), pp.1539-1543.

Soulsby, C., Birkel, C., Geris, J., Dick, J., Tunaley, C., Tetzlaff, D. 2015. Stream water age distributions controlled by storage dynamics and non-linear hydrologic connectivity: Modelling with high resolution isotope data. Water Resources Research, 51, 7759-7776.

Soulsby, C., Scheliga, B., Neill, A., Comte, J-C., Tetzlaff, D. 2021. A longer-term perspective on soil moisture, groundwater and stream flow response to the 2018 drought in an experimental catchment in the Scottish Highlands. Hydrological Prpceses , 35, e14206.

Spurgeon, D.J., Keith, A.M., Schmidt, O., Lammertsma, D.R. and Faber, J.H., 2013. Land-use and land-management change: relationships with earthworm and fungi communities and soil structural properties. BMC ecology, 13(1), pp.1-13.

Stirling, E., Fitzpatrick, R.W., Mosley, L.M. 2020. Drought effects on wet soils in inland wetlands and peatlands. Earth Science Reviews, 210, 103387.

Suriyagoda, L.D.B., Ryan, M.H., Renton, M., Lambers, H., 2014. Plant Responses to Limited Moisture and Phosphorus Availability: A Meta-Analysis. Advances in Agronomy, 124, 143-200.

Thomas, F.M. 2000. Vertical Rooting Patterns of Mature Quercus Trees Growing on Different Soil Types in Northern Germany. Plant Ecology, 147, 95-103.

Thompson, S.E., Harman, C.J., Heine, P. and Katul, G.G., 2010. Vegetation-infiltration relationships across climatic and soil type gradients. Journal of Geophysical Research: Biogeosciences, 115(G2).

Treseder, K.K., Schimel, J.P., Garcia, M.O., Whiteside, M.D. 2010. Slow turnover and production of fungal hyphae during a Californian dry season. Soil Biology and Biochemistry, 42, 1657-1660.

Turner, B.L. and Haygarth, P.M., 2003. Changes in bicarbonate-extractable inorganic and organic phosphorus by drying pasture soils. Soil Science Society of America Journal, 67, 344-350. https://doi.org/10.2136/sssaj2003.3440

Tye, A.M, Moir, A., Reinsch, S., Cartwright, C., Feeney, C.J., and Robinson, D.A., 2022. Environment and Rural Affairs Monitoring & Modelling Programme (ERAMMP). ERAMMP Report-70: Report on the use of remote sensing to assess soil erosion, poaching and disturbance features. Report to Welsh Government (Contract C210/2016/2017)

Unestam, T., 1991. Water repellency, mat formation, and leaf-stimulated growth of some ectomycorrhizal fungi. Mycorrhiza, 1(1), pp.13-20.

Van Genuchten, M.T., 1980. A closed-form equation for predicting the hydraulic conductivity of unsaturated soils. Soil science society of America journal, 44(5), pp.892-898.

Wilkinson, B., Broughton, W., Parker-Sutton, J., 1969. Survey of wind erosion on sandy soils in the East Midlands. Experimental husbandry, 18, 53-59.

William R. L. Anderegg, W, R.L., Berry, J.A., Smith, D.D., Sperry, J.S., Anderegg, L.D.L., Field, C.B. 2012. The roles of hydraulic and carbon stress in a widespread climate-induced forest die-off. PNAS, 109 (1) 233-237

Wouters, H., Keune, J., Petrova, I.Y., van Heerwaarden, C.C., Teuling, A.J., Pal, J.S., Vil-Guerau de Arellano, Miralles, D.G. 2022. Soil drought can mitigate heat stress thanks to a reduction of air humidity. Science Advances, 8, eabe6653.

Xu, Q.; Fu, H.; Zhu, B.; Hussain, H.A.; Zhang, K.; Tian, X.; Duan, M.; Xie, X.; Wang, L. 2021. Potassium Improves Drought Stress Tolerance in Plants by Affecting Root Morphology, Root Exudates, and Microbial Diversity. Metabolites, 11, 131.

Yang, F., Zhang, G.-L., Yang, J.-L., Li, D.-C., Zhao, Y.-G., Liu, F., … Yang, F. 2014. Organic matter controls of soil water retention in an alpine grassland and its significance for hydrological processes. Journal of Hydrology, 519, 3086– 3093.

Yang, Y.J., Liu, S.R., Wang, H., Chen, L., Lu, L., Cai, D.X. 2019. Reduction on throughfall reduces soil aggregate stability in two subtropical plantations. European Journal of Soil Science, 70, 301-310.

York, C. A., and P. M. Canaway. "Water repellent soils as they occur on UK golf greens." Journal of Hydrology 231 (2000): 126-133.

Yue, K., Yang, W., Peng, Y., Peng, C., Tan, B., Xu, Z., Zhang, L., Ni, X., Zhou, W., Wu, F., 2018. Individual and combined effects of multiple global change drivers on terrestrial phosphorus pools: A meta-analysis. Science of the Total Environment, 630, 181-188.

Zhang, Q., Shao, M., Jia, X., Zhang, C. 2017. Understory vegetation and drought effects on soil aggregate stability and aggregate -associated carbon in the loess plateau in China. Soil Science Society America Journal, 82, 106-114.

Zipper, S. C., M. E. Soylu, E. G. Booth, and S. P. Loheide II 2015. Untangling the effects of shallow groundwater and soil texture as drivers of subfield-scale yield variability, Water Resour.Res.,51, 6338–6358

Zeng, H., Tang, C-S., Cheng, Q., Zhu, C., Yin, L-Y., Shi, B. 2020. Drought-induced soil desiccation cracking behaviour with consideration of basal friction and layer thickness.

I: Review of the state of research on drought – Agricultural impacts

Ian P. Holman, Centre for Water, Environment and Development, Cranfield University

Overview

About the review

The variability in the UK's climate, topography and soils leads to multiple different outdoor and protected farming systems and large regional differences in their distribution. Agriculture is often cited as one of the most drought-sensitive sectors, with productivity and financial impacts developing over multiple time-scales and which can persist, materialise or be realised long after rain has returned and the drought has been considered to have ended. Because of these lags between the drought and some of the adverse consequences for farms and businesses, drought impacts may be ignored, or forgotten, in society and government.

What is covered: This review synthesises the state-of-the-art understanding of how agriculture and horticulture is affected by, and responds to, drought, sub-dividing the industry into (1) rainfed arable and livestock farming, (2) protected cropping and (3) outdoor irrigated cropping sub-sectors.

What is not covered: This review does not consider indoor livestock and poultry

Key findings

- Agricultural and horticultural sub-sectors are variably sensitive to meteorological, soil moisture, hydrological and (to a lesser extent) socio-economic drought.
- Drought impacts manifest in many different ways over a range of timescales, including through changes in productivity, quality, livestock welfare, business profitability and farmer wellbeing. Due to the resulting spatial complexity, there remains a poor granular understanding of drought impacts and the barriers and enablers of pro-active and reactive drought responses.
- The highly competitive financial environments facing most farmers leads to farm business decision-making strategies that increase the economic efficiency of production to the detriment of system redundancy, diversity and resilience to drought.
- Increasing drought resilience carries costs which do not deliver financial benefits in most years.

Recommendations

- Evaluating drought sensitivity and impacts across agriculture requires improved data.
- Better understanding of barriers and enablers for improving drought resilience is required, especially within irrigated production systems;
- The minimum lead time and forecast reliability that are required from drought monitoring and early warning systems to provide the business confidence to implement pro-active drought responses need to be determined;
- Future trends in the drivers affecting agriculture and agricultural water demand need to be better characterised through co-development with food supply chain actors. In particular, understanding the conditions in which investment in irrigation in currently rainfed livestock and arable farms becomes profitable, and the potential for future regional migration of UK irrigated production.

1. Introduction

Agriculture, as measured by the Utilised Agricultural Area (UAA), covers 71% or 17.2 million hectares of the UK, with 6.1 million hectares being croppable (Defra, 2021) and 99% being rainfed (Hess et al., 2020). The variability in the UK's climate, topography and soils leads to multiple different outdoor and protected farming systems and large regional differences in their distribution (Table 1). The South west and North West have 62% of the dairy industry which relies heavily on good grassland production for forage. The East of England contributes 30% of field vegetables, 35% of the potatoes, 62% of sugar beet and 28% wheat, 23% barley and 24% of the OSR. Whereas the South East contributes 43% of both top and small fruits.

Table 1. Regional percentage areas of key agricultural enterprises in England in 2021 (Adapted from Defra, 2022)

	Field	Pots	Protecte	HNS	Sugar	Top	Small	Wheat	Barley	OSR	Dairy	Beef	Sheep	Pigs
North East	2	1	0	0	0	0	0	4	5	6	1	8	13	3
North West	5	6	14	3	1	1	1	2	5	1	26	13	20	2
Yorkshire & Humber	18	19	13	3	8	0	1	14	16	16	7	12	14	38
East Midlands	26	15	9	26	24	2	4	20	16	19	6	11	8	9
West Midlands	7	15	12	8	2	30	21	10	7	11	14	13	14	5
Eastern	30	33	22	17	64	7	14	28	24	25	1	5	2	30
South East	6	3	25	11	0	44	50	13	13	14	5	9	8	5
South West	7	7	5	30	1	16	9	9	15	9	39	30	20	8

Notes: HNS Hardy nursery stock, Pots potatoes, OSR Oilseed rape)

Agriculture is often cited as one of the most drought-sensitive sectors, with the term "agricultural drought" describing low soil moisture content often used synonymously to mean "drought affecting agriculture". From a drought perspective, different farming systems have large differences in their sensitivity to the different types of drought (Table 2).

Drought impacts on agriculture develop over multiple time-scales and may persist long after rain has returned and the drought has been considered to have ended. These delayed impacts range from reduced crop yields and/or quality at harvest; shortages of stored fodder and bedding in the following winter; difficulty in filling on-farm reservoirs affecting irrigation supply in the following growing season; to reduced livestock fertility affecting births in the following year; to economic impacts on business profitability and viability (Figure 1). Because of these lags between the drought and some of the adverse consequences for farms and businesses, drought impacts may be ignored, or forgotten, in the wider community.

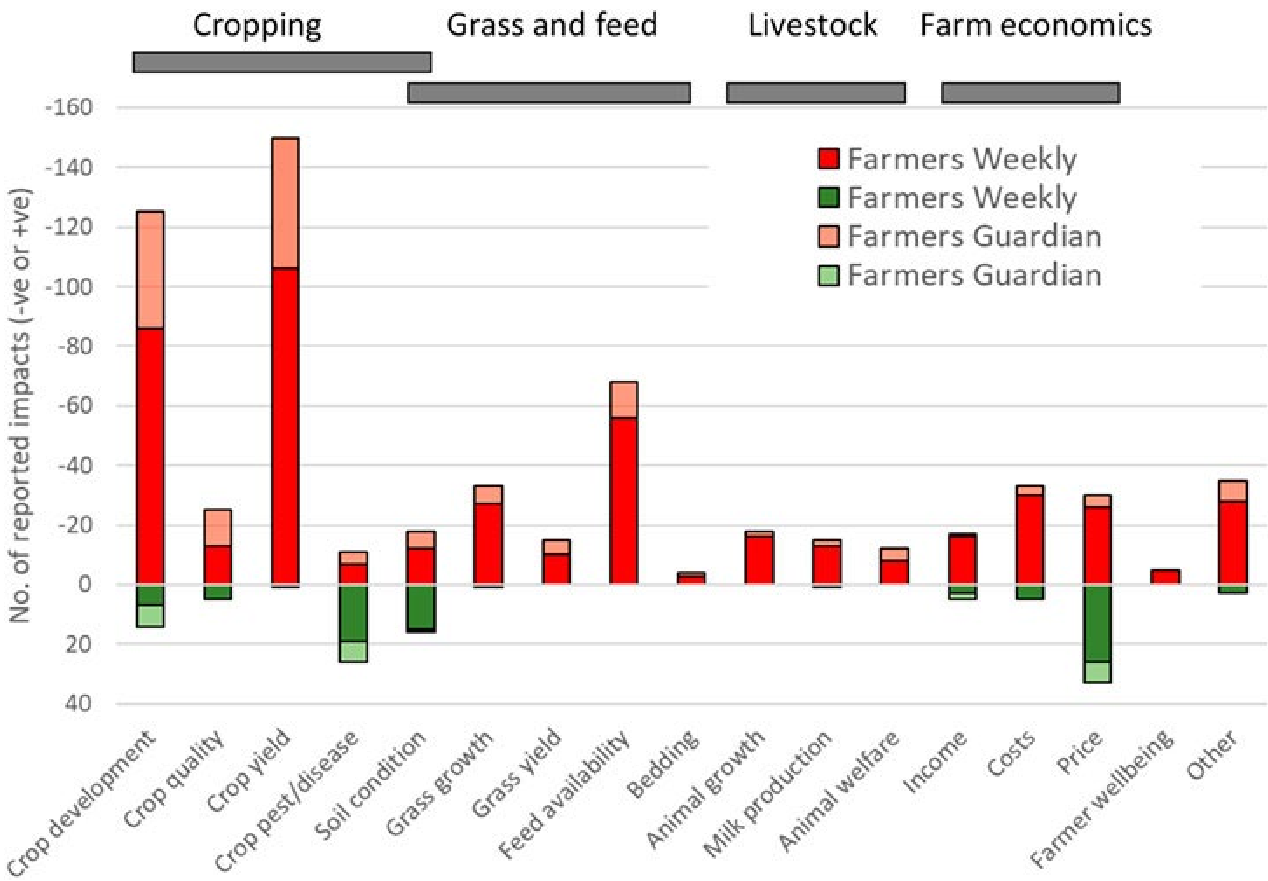


Figure 1. Negative (red) and positive (green) drought impacts reported in Farmers Weekly and Farmers Guardian trade magazines in 2018

Table 2. Sensitivity of farm types to different types of drought and water scarcity (adapted from Holman et al., 2019)

Farm type	Sensitive to:			
	Meteorological drought	**Agricultural drought**	**Hydrological drought**	**Socio-economic drought**
Outdoor livestock	×	✓ (lack of feed / fodder;	✓ (where reliant on springs and	✓ (fodder supply)

		increased costs)	streams for stock watering; abstractions for washing, cooling etc; and drainage ditches for wet fences)	
Arable (rainfed)	✗	✓	✗	
Arable and horticulture (irrigated)	✗	✓ (if licenced volume is insufficient)	✓ (due to restrictions on direct abstraction, but can be reduced by on-farm reservoirs)	✓ (irrigation equipment)
Protected cropping	✓ (where rainwater harvesting is employed)	✗	✓ (where reliant on direct abstraction or mains water, although certain exemptions exist)	
Indoor livestock and poultry	✗ (but may be sensitive to high temperatures)	✓ (lack of fodder/bedding; increased costs)	✗ (as animal welfare considerations likely to prevent restrictions)	

2. Review of drought impacts and adaptation in rainfed UK arable and livestock farming

Arable and livestock systems that rely on rainfall are the two dominant farming systems in the UK, but the UK's climate mean that they tend to be the most extensive farming types in the south / east and west/north of England, respectively.

There are much fewer drought studies related to livestock systems in the UK compared to crop-based systems, despite international understanding of the impacts of high temperatures and drought on grass yield and quality [e.g. Sheaffer et al., 1992]; animal productivity (quality and quantity of meat, milk, wool, etc.) and livestock fertility, particularly cows [e.g. De Rensis an Scaramuzzi, 2003; Alves et al., 2013].

2.1. Meteorological drought

Wreford and Topp (2020) suggested that rainwater harvesting and storage, which is sensitive to meteorological drought, may be used in some livestock farms but there were no cited studies.

2.2. Agricultural (soil moisture) drought impacts

All rain-fed agricultural systems in the UK are sensitive to the impacts of soil moisture drought as there is little opportunity for reactive adaptation (Rial-Lovera et al. 2018). However, the impacts on productivity (yield and quality: Keay et al., 2014; Knox et al., 2010) differ between drought events (due to severity and timing), between individual crops (including fodder crops) and between crop and grass varieties.

Wheat is less sensitive to drought than barley but has greater sensitivity that rye and triticale (Cho et al., 2012). Between about 25% (Foulkes et al. 2001) to 30% (Foulkes et al., 2007) of wheat in the UK is grown on shallow or sandy soils which are easily droughted. Yield losses from drought can range between 2 – 4.5t/ha (Foulkes et al., 2007), with modelled historic yield losses for var. Claire near Cambridge due to key drought events ranging from 15% in 1943 to 29% in 1976 and 38% in 1921 (Clarke et al., 2021). The importance of cultivar and drought timing was shown in a lysimeter study by Dickin and Wright (2008), who found that yields were reduced by drought from growth stage (GS) 61 by 17% (var. Xi-19) to 24% (var. Deben), whilst drought from GS 45 reduced yields from var. Claire by 53%.

Within crops, genetic improvements due to crop breeding and change in agronomy also makes estimating yield loss to drought problematic when separating natural variability from drought response (MacKay et al. 2011; Talbot 1984). Mechler et al., (2010) suggested that decreasing deviations from a 5-year moving average of yield during drought years in oilseed rape, unlike barley, was evidence of adaptation but may have reflected the most severe (1976) drought being at the beginning of the time series.

Notwithstanding impacts on yield, food crops have minimum standards that are required for their end market and drought during key growth phases may adversely affect the quality (OECD, 2023, NABIM, 2023) and thus price. However, there are few studies that assess drought impacts on crop quality (e.g. Rey et al., 2016).

The majority of studies looking at drought impacts in UK outdoor livestock systems have assessed the impacts of a soil moisture drought on fodder and forage crops. **Perennial ryegrass** (Lolium perenne L.) is the most important grass species in Britain, being a major constituent of both permanent and temporary grassland but is drought sensitive (Lambert et al., 2020). Grass growth during the 1976 drought was negligible in the south of England with many areas completely desiccated and animals needing supplementary feeding (Roy et al. (1978). Lee et al. (2019) found that drought caused a reduction in productivity in perennial ryegrass, but there was substantial variation between varieties (up to 82%), with the optimal variety changing depending on drought severity. Grass yields in dry years can be increased through inclusion of other grasses (Lambert et al., 2020) and legumes (Kuchenmesiter et al., 2013). **Forage maize** (Zea Mays L.) is an increasingly important fodder crop for intensive production systems, which requires 300-400mm from May to October in Southern England (Bunting, 1978) and is suggested to have medium-high sensitivity to drought (Brouwer & Heibloem, 1986) but good response to irrigation (Klocke et al. 2007; 2014).

Agricultural practices can increase the drought sensitivity of grassland, with reductions in soil water retention in improved grassland compared to neighbouring semi-natural grassland being ascribed to soil compaction (Wallace and Chappell 2020), which is common in some grassland systems (Palmer and Smith, 2013). Irrigation is currently not commonly used in grassland systems in the UK to reduce drought sensitivity (Wreford and Topp 2019), but the impacts of drought on permanent pasture in Wallace and Chappell (2020) were offset by the addition of moisture from summer slurry applications which enabled continued sward growth and reduced sward damage.

There are very few studies that look at the indirect impacts of drought on livestock and meat and milk production. Salmoral et al. (2020), combining analysis of UK national farmers magazines and farmer interviews in Derbyshire, showed that the reported impacts of drought were overwhelmingly negative and spanned pasture growth and yields, feed and bedding availability, animal growth and welfare, milk production, financial performance and farmer wellbeing.

Wreford and Adger (2010) analysed the deviation from the five-year moving average around historic drought years in UK cattle, sheep, pig and poultry production, and showed that sheep and cattle production tended to both increase and decrease whilst pig and poultry production mostly decreased. However, they recognised that their method could not absolutely separate the effects of drought, policy changes and farmer responses. Mechler et al., (2010) similarly found livestock production to have a varying response to drought/heat wave years over time and that it is difficult to unpick the effects of weather on livestock production from policy or other shocks. Similarly, in their study of heat stress impacts on milk yields, Dunn et al. (2014) concluded that their observed changes could be the combined effects of reduced pasture quality due to drought associated with the hot weather and heat stress in the animals.

2.3. Hydrological drought impacts

Livestock farms require access to reliable supplies of water as animals need a regular daily water supply to regulate body temperature and maintain organ functions (Ward & McHague, 2007). Drinking water for livestock is the largest agricultural use of water accounting for 41% of total use or 75 million m3, with a further 13 million m3 used for washing down (Defra 2011). Consequently, they can be highly reliant on private boreholes and small water sources such as drainage channels, streams and springs to provide drinking water for cattle and sheep (Salmoral et al., 2020; Hess et al., 2018) and to form ‘wet fences’ to control livestock movement (Holman et al., 2019). The proportion of farms abstracting water from surface water for immediate use is significantly higher in DA (about 53%) and SDA (about 69%) farms compared to non-LFA farms (about 25%) (Defra 2017), although no volumes were recorded. Consequently, LFA grazing livestock farms used the lowest percentage of mains water of all farms types (Defra 2017). Given that between 25-39% of sheep, beef cows and other cattle have access to watercourses (Defra, 2011), the drying up of such small water sources during droughts can lead to significant costs and effort in order to provide livestock with vital alternative water supplies. Riley et al. (2018) consider that the high connectivity of small water bodies with their surrounding environment makes them particularly vulnerable to having highly modified discharge as a consequences of agricultural and forestry drainage, with implications for flood and drought control further downstream but make no mention of the associated hydrological drought risk to livestock agriculture. Consequently, the financial implications of the short-term failure of small water supplies during hydrological drought, particularly in upland areas which could lead to a reduction in herd size through slaughter or sale of stock (Hess et al., 2018), are unknown.

2.4. Drought responses

Research in the UK has shown similarities in reported drought responses within outdoor livestock farming at national and local scales that are largely reactive, crisis-driven and contribute to the hidden personal burden of extreme events on farmers (Salmoral et al., 2020). The main farm-level coping strategies reported from the 2018 drought were: (1) management of available grazing and feed; (2) selling livestock to reduce feed demands and improve cash flow and (3) purchase of additional feed (Salmoral et al., 2020). Additional short-term measures included Government derogations or temporary prescription adjustments (TPA) to relax certain regulations on farming activity; and private sector actions including flexible loan agreements from financial institutions; and modifications to retailers' and processors' contractual terms and specifications of livestock products (Salmoral et al., 2020). Feed purchases in 2018 were affected by socio-economic drought impacts as fodder import subsidies in the Republic of Ireland increased competition for domestic feed (Byrne, 2018).

Wreford and Adger (2019) identify many longer-term adaptation options to variable precipitation and water availability for the livestock sector. For example, drought impacts may be reduced by introducing drought tolerant species (e.g. Marshall et al., 2016; Neal et al., 2009), incorporating legume and herb forage species that provide greater nutrition into pastoral systems (e.g. Kemp et al., 2010) or general species diversification (Kirwan et al., 2007), but consideration needs to be given to growth patterns and production trade-offs (Lee et al., 2013; Wreford and Topp, 2020). However, there is scant evidence of increased drought resilience resulting from adaptation within the outdoor livestock sector (Salmoral et al., 2020; Wreford et al. 2010). The majority of farmers interviewed in Salmoral et al. (2020) had no measures to reduce the impact of future droughts. Wreford et al. (2010) suggested declining production impacts of recuring UK droughts were indicative of evidence of adaptation within pigs and poultry production, although the national scale of analysis doesn't allow for the differing regional nature of the droughts or balance of indoor versus outdoor production systems. Wreford et al. (2010) did not look at dairy production due to the impacts of changing policy on production.

3. Protected cropping

3.1. Background

Protected cropping can include both edible and non-edible produce and relates to crops which are grown under polythene or glass. The area of protected cropping under glasshouses and polytunnels (including Spanish polytunnels) has increased greatly in recent years, particularly in the soft fruit sector. In the last Glasshouse survey (Defra 2007), 22% of the total protected area was under polythene but this is expected to have increased greatly in area and proportion since. There are currently around 800 ha of protected fresh vegetables, worth £374M, and 3000 ha of glasshouse crops (Defra 2021). However, an unknown proportion of the 12,000 ha of plant and flower production, worth £1,137M, and 10,000ha of soft fruit worth £629M (Defra 2021), is grown under protection. However, 81% of Hardy Nursey Stock growers used irrigation, based on the 2005 irrigation survey (Knox et al., 2010). Earlier estimates of the proportion of strawberries (the main soft fruit) grown under polytunnels include grower estimates of at least 70% (Evans 2013); whilst Calleja et al. (2012) estimate 81.3% of the strawberry crop on 'large farm enterprises' to be under polytunnels in 2009. Defra (2011) estimated that 9+2 % of farms use rainwater storage but no indication of use is provided, although irrigation using collected rainwater was estimated at 442,000 m3 in 2010. Mains water supply, which makes up about 1% of total irrigation water for agriculture, is mostly used on protected edibles cropping to ensure reliability of supply and microbiological water quality (Hess et al., 2020).

3.2. Impacts of drought

Salad crops are increasingly grown hydroponically using artificial substrates or nutrient film technique (NFT), whilst the UK soft fruit and ornamental horticultural industry are increasingly using container grown systems, including bags, buckets, troughs and modules (Atwood, 2013). The underlying drought sensitivity of protected cropping stems from their complete reliability on irrigation and, particularly in container or pot-base production, the smaller soil volume available for water storage (Lea-Cox et al., 2001).

These are all high value sectors that are entirely reliant on irrigation to maintain yields and quality (Rey et al., 2018). Increasing pressures to reduce the volume of water used due to rising costs of mains water, reduced availability of water in some catchments and changes to abstraction licencing combined with concerns regarding runoff and enhanced downstream flood risk (Entec 2006, Centre for Rural Research 2008), has led to growing interest in rainwater harvesting. This can be as a sole water source, or in combination with surfacewater abstraction, groundwater abstraction or mains water supplies (Dunn and Adlam 2019).

Protected cropping can therefore potentially be vulnerable to meteorological drought when rainfall supplies (frequency and amount) are insufficient for rainwater harvesting to meet irrigation needs; and hydrological drought if abstraction licences are unable to meet irrigation need. Although crops grown under protection and container-grown crops have an exemption from Section 57 emergency drought restriction on surface water abstraction (AHDB, 2022), abstraction licences for irrigation of protected cropping can be subject to hands-off flow conditions. However, due to the high financial value of protected crops (Rey et al., 2018) and their very high sensitivity to water stress, such businesses are likely to have back-up from groundwater and/or mains water. There is a statutory excemption from Drought Order restrictions for irrigating plants that are grown or kept for sale or commercial use using mains water, reducing the likelihood of supply restrictions. Consequently, the direct impacts of drought are likely to be felt through reduced product quality and increased production costs, but there is little information on drought impacts on this sub-sector.

4. Outdoor irrigated cropping

4.1. Background

Unlike many cereal crops which are never seen by the consumer, the value of fresh produce is strongly determined by visual aspects and buyer protocols which include size, shape and skin finish (Hingley et al., 2006, OECD, 2018). Consequently, controlling soil moisture through supplemental irrigation becomes critical for growing high quality, high value fresh produce in drier regions of the country.

Irrigation was second largest user of water in agriculture accounting for 38% of the total volume used or 70 million m3 (Defra 2011). Main crop potatoes and vegetables for human consumption dominate the irrigated area (38% and 25%, respectively) and volume applied for irrigation (48% and 26%) in 2010 (Defra 2011). UK potato and outdoor vegetable production were worth £820M and £1320M, respectively, in 2020 (Defra, 2020), although not all production is irrigated. High-value irrigated vegetable cropping typically abstracts 160 × 106 m3 in a dry year (Weatherhead et al., 2015) with half the total irrigated area and 57% of the total volume of water applied concentrated in the Anglian region

4.2. Impacts of hydrological drought

The emphasis of retailers' on quality standards within the fresh produce supply chain has resulted in supplementary irrigation becoming essential to ensure the viability and profitability of key crops, including potatoes, outdoor vegetables and orchard fruit, in some regions. Rey et al. (2016) estimated that the on-farm financial net-benefits of irrigation for these outdoor crops that would be lost in a 'design' dry year if irrigation in England Wales were banned were around £204 million, reflecting the crop-specific loss of quality and yield benefits. Anglian Water and University of Cambridge, (2013) reported that drought in 2011 led to an estimated 15% reduction in potato yield.

However, abstraction of water for agricultural irrigation has the lowest priority for water allocation in the UK under drought conditions in order to protect public water supplies and the environment (Salmoral et al, 2019). Salmoral et al (2018) probabilistically estimated the national- and regional-scale economic impacts arising from mandatory abstraction restrictions (section 57 restrictions) within drought management plans (taking account of catchment-specific river flows and irrigated crop areas). However, these impacts do not reflect the impacts arising from irrigation constraints imposed by Hands Off flows as abstraction licence conditions.

4.3. Responses to drought

Agriculture's low priority for water during drought partly reflects perceptions that the financial value of water use in agriculture and horticulture is low compared to other sectors (despite high financial benefits per unit of water applied for some crops - Rey et al. 2016), and that there is scope to increase the 'efficiency of use' of agricultural irrigation during drought conditions. Whilst irrigating at night to reduce evaporation losses was the most common short-term coping strategy identified by Rey et al., (2017), changing irrigation schedules to either reduce irrigation depth over the full area or reduce the area irrigated were also common strategies (Knox et al., 2000)- both of which lead to (uncertain) yield and quality penalties. Socio-economic drought impacts also hindered responses in 2018, when hiring or buying additional irrigation equipment was challenging due to a lack of equipment on the second-hand market and limited supplies of new irrigation equipment due to the wider European drought (National Farmers Union, 2019)

In common with many regions or countries, the allocation of water for agricultural irrigation remains economically sub-optimal. The lack of regulatory flexibility to overcome the historic limitations of licence allocations (e.g. those awarded on a first come, first served basis) means that many irrigation licences have allocated water that is not abstracted, even in drought years (Chengot et al., 2021); whilst other abstraction licences have been fully utilised and have unmet demand. Multiple bureaucratic, financial and timing constraints on water trading and contract options (Rey et al., 2019; Rey, 2015) limits their utility to address this mismatch in the UK.

Longer term actions reported in Rey et al. (2017) included the development of a farm drought management plan to inform drought responses (76% of respondents), investment in alternative water resources and more efficient irrigation infrastructure (43%) and modification of crop choice and planting programmes (18%) (Rey et al., 2017). There has been major investment in on-farm reservoirs to synchronise the timing of abstraction with water availability rather than irrigation need (NFU 2015; Rey et al., 2017), which may partly explain the perceived improvements in drought resilience within irrigated growers in easter England (Rey et al., 2017).

5. Discussion of knowledge gaps

Given the complexity of the UKs agricultural systems and their differing sensitivity to droughts outlined above, it is inevitable that there are many uncertainties in our understanding. Key issues emerge from the above studies relating to:

- Understanding the impacts arising from past droughts
- Barriers and enablers for improving drought resilience
- The potential of drought monitoring and early warning systems
- Future trends affecting agriculture and agricultural water demand

5.1. Understanding past and current impacts

Despite agriculture's widely acknowledged sensitivity to drought, there remains poor quantitative and empirical evidence of the impact of drought on UK farming. This goes from drought impacts on the production (yields and, especially, quality) of crops, livestock and livestock outputs through to the financial costs to farm businesses, the supply chain and consumers. Consequently, there is a lack of baseline evidence to underpin operational and regulatory decisions and to demonstrate the need to pro-actively support agriculture and food supply chains to become more drought resilient (Table 3). This partly contributes to the largely reactive, short-term responses from regulators, farmers and the food industry that are seen during each drought event.

The impacts of each drought are different due to the interaction of drought event characteristics (location and spatial extent; timing and severity) with agriculture and horticulture's drought vulnerability which differs regionally, between farming systems and between farms due to different soils, water sources, crops, livestock and markets. Modelling assessments provide simplified insights into potential crop-specific yield losses due to drought but, for example, only one wheat cultivar on the AHDB recommended list to growers had been calibrated in the widely used Sirius wheat model (Clarke et al., 2021) that was estimated to be grown on 2% of the total UK wheat area.

The complex inter- and intra-farm differences mean that understanding the financial impacts of drought is challenging as multiple effects make up the aggregate drought impact, including drought-induced changes in productivity, disease burden, input costs and output prices (Salmoral et al; Clarke et al., 2021). All of these can change positively or negatively as a consequence of a drought, leading to a spatially complex mix of winners and losers at all scales e.g. crop yields can be adversely impacted by soil moisture stress or be increased due to higher radiation levels where soil moisture stress is low due to soil type, marginal climatic areas or due to irrigation. Consequently, regional droughts do not clearly affect national yields (Clarke et al., 2021) but have significant regional economic damages.

There are few robust assessments of the regional or national impacts of drought on crop production and financial damages in the UK. It is difficult to separate the impact of soil moisture drought from restrictions on irrigation water abstractions during hydrological drought, but dry weather between 2010 and 2012 was estimated to have caused losses of £400 million to UK agriculture (Anglian Water, University of Cambridge. 2013) with losses in 2012 valued at £72 million to irrigated potato production in England (Akande et al., 2013). Knox et al. (2000), Rey et al. (2018) and Salmoral (2018) have all assessed the potential financial impacts of restrictions on abstraction, based on integrating the severity and timing of restrictions, irrigating cropping, crop prices and the crop-specific yield and crop quality benefits arising from supplemental irrigation that would be lost. However, this latter component of the methods are based on Morris et al. (1997) and may no longer adequately reflect benefits.

Outdoor vegetable growers maintain a buffer or 'headroom' between the expected dry-year irrigation need of their cropping plan (assumed to be based on the 16th driest year in 20- Weatherhead et al., 2002) and their water resources allocation (licenced volume). This aims to reduce the likelihood that a volumetric shortage during a drought will reduce their output of marketable produce and/ or lead to financial loss. Consequently, most irrigation abstraction licences are not fully used in most years, leading to a perception of an inefficient allocation of water. However, there are other legitimate agronomic reasons for having headroom, even in dry years, which need to be understood. Otherwise there is a risk that reducing licence volumes to reduce (apparent) headroom will lead to reduced crop production and/or increased risk of financial penalties to growers and increased costs through the food supply chain during drought.

Not all hydrological drought impacts on irrigated agriculture and horticulture arise from abstraction restrictions imposed through Drought Plans i.e. Section 57 restrictions. The inability to meet crop irrigation needs in a given year can also arise from the irrigation need being greater than the design dry-year, Hands Off Flow conditions stipulated within abstraction licence conditions, insufficient capacity in the irrigation system design (in pumping, conveyance and application) to meet peak irrigation need and insufficient reservoir storage. The relative significance of these is uncertain.

Table 3. Key knowledge gaps in understand drought impacts on agriculture

System	Data gap
Outdoor livestock	What is the frequency and severity of soil compaction, given it reduces soil water holding capacity and sensitivity to meteorological drought?
	How resilient are small upland and lowland waterways to hydrological drought? Enabling better understanding of hydrological drought impacts in grassland areas; and potential stresses on rural mains water supplies?
Protected cropping	What is the current areas of polytunnels, given that 100% of crop water need is met by irrigation? Permanent and seasonal polytunnels (as opposed to glasshouses) widely associated with the production of soft fruit and ornamentals are not captured by Ordnance Survey mapping products
	What sources of water does protected cropping use, including rainwater harvesting?
	How common are reservoirs within protected cropping businesses and what is their storage capacity?
	What are the trends in the use of mains water in protected cropping?
Irrigated agriculture	What are the areas and geographical distribution of irrigated crops? There are no robust datasets of the area and distribution of individual irrigated crop types. GIS products such as Land Cover Map Plus and CROME do not separate rainfed from irrigated crops.
	What is the frequency and area of different irrigation application methods and scheduling approaches. Methods have potentially evolved but the most up-to-date results are from the detailed water module within the 2009/10 Farm Business Survey (FBS) and the last (2010) Irrigation survey were published in Defra (2011).
	What is the reservoir storage capacity of irrigated growers? Whilst there are data on storage licence limits, the relationship between this and reservoir capacity is unknown
	Are irrigators still designing for a historic 4 in 20 dry year?

	What are the reasons (and their relative importance) for headroom and its annual variability beyond weather? i.e. as a safety net for dry years; having insufficient irrigated land; rotational limits of irrigated crops; land tenure (land rented out to bigger irrigators) etc
	What are the economic and environmental costs and benefits of Section 57 restrictions to provide an evidence-based justification for mandatory Section 57 restrictions on agriculture and horticulture

5.2. Understanding drought responses

Increasing the resilience of agricultural systems to drought can be achieved by the farmer reducing the probability of the drought event, increasing the robustness of the system to drought or facilitating recovery (Hess et al., 2018). Probability can be reduced by low cost options such as improving the water holding capacity of soils (soil moisture drought) or high cost investments such as reservoir storage (hydrological drought)- however, large scale investment can lock growers into their current production system and lead to reduced flexibility. Robustness is generally achieved through building diversity (having access to different water sources; growing a greater range of crops; being able to substitute grazing for conserved fodder (Salmoral et al., 2020) and/or redundancy (e.g. having under-utilised grazing: Salmoral et al., 2020; spare irrigation equipment, licence headroom). However, there is a tendency to reduce diversity and redundancy, as (1) market pressures drive increased economic efficiency, economies of scale and reduced marginal costs of production (Abson et al., 2013); (2) regulatory pressures are moving towards reducing headroom; and (3) both diversity and redundancy have opportunity costs (Abson et al., 2013) that, unless outweighed by the avoided penalties of infrequent droughts, lead to the 'spare' resources being utilised. Consequently, the highly competitive financial environments facing most farmers have a tendency to lead to farm business decision-making strategies that increase the economic efficiency of production to the detriment of system redundancy, diversity and resilience to drought (Hess et al., 2018). This leads to a trade-off between the high expected gross margins (a common indicator of the economic performance of farm enterprises) of specialised production systems and the expected variance of those margins as a consequence of variability in environmental, economic, and policy perturbations (Abson et al., 2013). Consequently the current and future land use choices are, and will be, influenced by the risk to returns preferences of individual farmers (Baumgärtner et al., 2010; Di Falco and Chavas, 2006) and their perceptions of drought risk.

Increasing resilience to drought carries costs, including capital investment costs (e.g. farm reservoir construction), operational costs (e.g. soil management) and/or profit-foregone (e.g. reduced crop area), so that there will not be financial benefits of change in all years. Consequently, different typologies of farms will have different constraints and risk appetite – farms with limited financial capacity to adapt may be more likely to adopt practices that save money in all years (Rial-Lovera et al., 2017); whilst capital intensive production systems may select risk averse options that minimise regret (Knox et al., 2010).

Increasing social capital has been shown as a low-cost means of enhancing drought resilience (Holman et al., 2021). As there are thousands of farms, both large and small, dependent on supplemental irrigation, irrigators arguably need to have a more coordinated and coherent voice at the local level. This can be facilitated by the formation of Water Abstractor Groups (WAGs) (Whaley and Weatherhead, 2015a) that fosters trust (between farmers and between farmers and the regulator; Whaley and Weatherhead 2015b), increased negotiating power (Leathes et al., 2008) and increased adaptive capacity through improved social networks and knowledge exchange (Holman and Trawick, 2011). However, despite the acknowledged benefits found, there remain only a limited number of active WAGs (Whaley and Weatherhead, 2015c).

Collaboration also has the potential to deliver further benefits in irrigated agriculture through facilitating the collaborative management of licenced water resources (Whaley et al., 2015b) through water trading (Rey et al., 2019), secondary water markets (Rey 2015) and water sharing (Chengot et al., 2021) to support more efficient water allocation. Chengot et al. (2021) showed that water sharing could potentially significantly reduce irrigation deficits during drought across a group of abstractors in an Eastern England catchment within the pre-existing abstraction licence and HoF constraints that prevent environmental degradation. However, there remains a lack of understanding around the potential operationalisation of such approaches (Rey et al., 2021).

Table 4. Key knowledge gaps regarding agricultural responses to drought

Sub-sector	Knowledge gaps
Irrigated cropping	What is the potential for no-regret reductions in irrigation water abstraction during drought?
	What are the enablers and barriers for investment in more efficient irrigation application systems and scheduling systems?
	What is an appropriate level of current dry-year headroom to provide resilience for future droughts?
	What is preventing the wider development of Water Abstractor Groups?

	What is the potential of secondary water markets and water sharing to allow agriculture to make better use of its allocated water? More flexible and collaborative approaches to irrigation abstraction have the potential to enable better water efficiency but the implications of regulatory barriers, HoFs and s57 restrictions are not sufficiently understood
	How do retailer contracts limit (or enable) responses to hydrological drought and change the financial consequences of abstraction restrictions?

5.3. Drought forecasting

The reactive responses of most UK farmers to drought (Salmoral et al., 2017; Rey et al., 2017) is partly due to a lack of reliable information on the state and expected changes in environmental conditions. Drought monitoring and early warning (MEW) systems offer the promise of enabling appropriate and timely management actions (Hannaford et al., 2019). They typically use drought indicators (WMO and GWP 2016) to monitor the status of rainfall, river flows, groundwater levels, and other hydrometeorological variables, relative to historical conditions, rather than impact indicators (Bachmair et al., 2016).

Key issues (Table 5) for the design of a useful MEW for agriculture:

- **The choice of indicators:** it is unclear whether standardised indicators (such as the SPI used in the UK Drought portal), quantitative multivariate composite indicators (as recommended by Hannaford et al. 2019) or impact indicators (Clarke et al 2021) will have resonance to the diverse agricultural and horticultural industry. Clarke et al (2021) found that the correlation between SPI and SPEI with simulated yield loss were similar, suggesting that SPI may suitable for UK drought monitoring for wheat. However, evapotranspiration rates are expected to increase due to climate change, so that SPEI may be more appropriate for longer-term use (Haro-Monteagudo et al., 2017). Reinforcing this, Parsons et al. (2018) found that a generalised linear model using SPEI-6 gave better results against reported impact data. Standardised indicators cannot be interpreted reliably at a national scale in a country with as much regional variation in climate and agricultural practice as the UK. The same value of SPEI in different locations corresponds to very different soil moisture deficits and growing conditions (and thus soil moisture drought severity), but Parsons et al. (2018) found that this did not fully explain observed regional differences in reported drought impacts. They suggested that such standardised indicators would need to be calibrated on a regional (or smaller) scale to be used as an indicator of agricultural drought within a MEW system.

- **Forecast lead times:** a business-appropriate forecast lead time is needed to enable farmers to take pre-emptive action to minimise the financial impact of drought. However, these lead times are likely to be different in different agricultural production systems e.g. informing buying in additional livestock feed during the early stages of a drought compared to changing the crop or variety selected for planting
- **Forecast reliability:** Given the reliability of forecasts with sufficient lead-time to inform decision and the relative infrequency of drought risk, farmers are reluctant to make changes that they know will lead to reduced returns if a drought does not occur (Rial-Lovera et al., 2017; Hess et al., 2018). Over 76% of potato growers in Rey et al., (2023) had forward contracts for between 50-100% of their potato crops, with over 50% of contracts specifying financial penalties if the grower were unable to fully deliver due to a water-related issue.

Table 5. Key knowledge gaps related to MEW systems for agriculture

Questions
How have farmers and growers modified their decisions (if at all) based on currently available MEW outputs?
What drought or drought impact indicators are meaningful to the agricultural and horticultural sector?
What MEW lead times are needed to inform decisions in different agricultural sub-sectors?
What forecast reliability is needed to provide the confidence to implement pro-active drought responses?

5.4. Understanding future trends

Following a long period of increasing abstraction, Weatherhead et al. (2015) reported that the volume of water abstracted for irrigation was declining by -1.4%/yr between 1990-2010 after allowing for annual weather variability, partly reflecting increased yields and hence decreasing crop areas, increased efficiency, better irrigation scheduling and reduced water availability and reliability. However, it is inappropriate to continue to project such past trends forward due to potential future changes in agroeconomic policy, market needs, agroclimate variability and change, and resources availability (Knox et al. 2018) (Table 6).

Agricultural water demand forecasting is essential to informing future policy and strategies (e.g. Knox et al., 2013; Weatherhead et al., 2015). There have been numerous published studies of the impacts of climate change on irrigated agriculture including the implications for the (unconstrained) irrigation needs of currently irrigated crops (e.g. Weatherhead and Knox, 1999; Dacchache et al. 2011) and potential changes in the spatial distribution of irrigation demand due to changing climate suitability (e.g. Dacchache et al. 2012). However, there are multiple barriers (Knox et al., 2010) to transformational changes in farming and food systems, such as movements in key commodity production areas in the UK, in response to more extreme conditions (Rickards et al., 2012; Folke et al., 2010).

Supermarkets have increasingly dominated the UK grocery market so that their focus on product specification (aesthetic, size and quality standards) has driven supplemental irrigation practice and investment (Knox and Hess, 2018; Sutcliffe et al., 2021; Rey et al; 2023). Widespread use of irrigation depends on the profitability of irrigation, which in turn depends considerably on the price differentials offered for quality produce in the market (Knox et al., 2007). The combination of the challenges of meeting product specifications within forward contracts during drought years; price competition with discount supermarkets and the reducing profitability of fresh produce may lead to changes in markets (pre-pack versus processing) and irrigation practice.

There is much less understanding of the increased future water requirements for currently rainfed crops and livestock systems. The main adaptations to drought within rainfed cropping include earlier planting, selection of more drought tolerant crops and varieties cultivars, changed cultivation practices, and increasing soil organic matter (Rial-Lovera et al., 2017). As pro-active measures, they cannot reduce impacts once a drought has started, unlike supplemental irrigation. Irrigation of wheat in the UK has traditionally been considered uneconomic, but increases in world wheat prices, increased drought frequency and the impacts of climate change are likely to make the financial benefits stronger (El Chami et al., 2015). Similarly, given observed drought impacts on maize yield in 2018 and the rapid expansion of anaerobic digestion plants, supplemental irrigation of maize may become more common (Knox and Hess, 2019).

Irrigation is currently not commonly used in livestock systems in the UK. Future changes in drought and/or climate that reduce growing season rainfall will reduce the quantity and quality of grass forage produced and may (where abstraction water is available) increase the adoption of supplemental irrigation in intensive grassland systems (Wreford and Topp, 2020) to reduce the need for supplementary feeding (Roy et al. (1978).

However, for most currently rainfed crops and on most soils, supplementary irrigation will not be needed in most years. Consequently, there is a need to better understand the regions and conditions in which rainfed farms would invest in supplementary irrigation (leading to increased demand for irrigation abstraction) and the situations within which irrigated farms would increase irrigation of the currently rainfed crops in their rotations (leading to increased usage of their volumetric limits).

6. Conclusions

This review has shown that the sensitivity of farming to drought differs between agricultural and horticultural sub-sectors, due to their reliance on different sources of water, so that agricultural and horticultural sub-sectors are variably sensitive to meteorological, soil moisture (agricultural) and hydrological droughts (and to some extent socio-economic drought). Drought impacts manifest in many different ways over a range of timescales, including through changes in productivity, quality, livestock welfare, business profitability and farmer wellbeing. Due to the resulting spatial complexity, there remains a poor granular understanding of drought impacts and the barriers and enablers of pro-active and reactive drought responses.

References

Abson, D.J.; Fraser, E.D.G.; Benton, T.G. Landscape diversity and the resilience of agricultural returns: A portfolio analysis of land-use patterns and economic returns from lowland agriculture. Agric. Food Secur. 2013, 2

AHDB (2022). Protecting the water supply for your crops. https://ahdb.org.uk/knowledge-library/protecting-the-water-supply-for-your-crops

Akande, K.; Hussain, S.; Knox, J.; Hess, T.; Hooftman, D.; Stratford, C.; Schafer, S.; Acreman, M.; Edwards, F. The Impacts of Drought in England; R&D Technical Report WT0987/TR; Department for Environment, Food and Rural Affairs Water Availability Division: London, UK, 2013.

Alves, M.F., Gonçalves, R.F., Pavão, D.L., Palazzi, E.G., Souza, F., Queiróz, R.K.R.; de Achilles, M.A. Effect of heat stress on the maturation, fertilization and development rates of in vitro produced bovine embryos. Open Journal of Animal Sciences 2013, 3(3): 174–178.

Anglian Water, University of Cambridge. 2013. Water, water everywhere? Encouraging collaborating and building partnerships. Institute for Sustainable Leadership, University of Cambridge. https://www.cisl.cam.ac.uk/business-action/business-nature/natural-capital-impact-group/pdfs/water-water-everywhere-scroll.pdf

Atwood J (2013). Water harvesting and recycling in soft fruit. Availablw from https://horticulture.ahdb.org.uk/knowledge-library/water-harvesting-and-recycling-in-soft-fruit

Bachmair, S., Stahl S, Collins K, Hannaford J et al, 2016a: Drought indicators revisited: The need for a wider consideration of environment and society. Wiley Interdiscip. Rev.: Water, 3, 516–536, https://doi.org/10.1002/wat2.1154

Baumgärtner S, Quaas MF: Managing increasing environmental risks through agrobiodiversity and agrienvironmental policies. Agr Econ. 2010, 41: 483-496. 10.1111/j.1574-0862.2010.00460.x.

Brouwer C and Heibloem. 1986. Irrigation water management: Irrigation water needs. FAO Training manual no. 3. Rome: FAO.

Brown C, Alexander P, Almuth A, Holman IP, Rounsevell MDA (2019). Achievement of Paris climate goals unlikely due to time lags in the land system. Nature Climate Change 9(3): 203-208

Bunting ES. 1978. Maize in Europe. In: Bunting ES, Pin BF, Phipps RH, Wilkinson JM and Gunn RE (Eds.). Forage Maize. Agricultural Research Council: London. 16 – 21

Byrne J (2018). Ireland importing fodder to tackle crisis. https://www.feednavigator.com/Article/2018/04/09/Ireland-importing-fodder-to-tackle-crisis (accessed 06/03/23)

Calleja E, Ilbery B, Mills P (2012). Agricultural change and the rise of the British strawberry industry, 1920–2009. Journal of Rural Studies, 28: 603-611

Centre for Rural Research (2008). Polytunnels and rural communities. Report for the Commission for Rural Communities. Available from https://eprints.worc.ac.uk/7363/1/Polytunnels%20Report%20Final%2031-03-08.pdf

Chengot R, Knox JW, Holman IP (2021). Evaluating the Feasibility of Water Sharing as a Drought Risk Management Tool for Irrigated Agriculture. Sustainability, 13, 1456. https://doi.org/10.3390/su13031456

Cho, K.; Falloon, P.; Gornall, J.; Betts, R.; Clark, R. Winter wheat yields in the UK: Uncertainties in climate and management impacts. Clim. Res. 2012, 54, 49–68

Clarke D, Hess TM, Haro-Monteagudo D, Semenov MA, Knox JW (2021). Assessing future drought risks and wheat yield losses in England. Agricultural and Forest Meteorology 297: 108248

Daccache, A., Keay, C., Jones, R.J.A., Weatherhead, E.K, Stalham, M.A., and Knox, J.W. (2012), 'Climate change and land suitability for potato production in England and Wales: impacts and adaptation', Journal of Agricultural Science, Vol 150, No 2, pp161–177.

Daccache, A., Weatherhead, E.K., Stalham, M.A., and Knox, J.W.(2011), 'Impacts of climate change on irrigated potato production in a humid climate', Agricultural and Forest Meteorology, Vol151, pp 1641–1653

Defra (2007). Glasshouse Survey England. Available at https://www.data.gov.uk/dataset/0b0a26bc-236d-471b-86c9-47557c32077f/glasshouse-survey-england

Defra (2011). Water Usage in Agriculture and Horticulture: Results from the Farm Business Survey 2009/10 and the Irrigation Survey 2010. Available from https://webarchive.nationalarchives.gov.uk/ukgwa/20130123162956/http:/www.defra.gov.uk/statistics/files/defra-stats-foodfarm-farmmanage-fbs-waterusage20110609.pdf

Defra (2017). Water usage on farms: Results from the Farm Business Survey, England 2015/16. Available from https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/587296/fbs-wateruse-statsnotice-31jan16.pdf

Defra (2021). Agriculture in the United Kingdom 2021. Available from https://www.gov.uk/government/statistics/agriculture-in-the-united-kingdom-2021

Defra (2022). Structure of the agricultural industry in England and the UK at June: regional. Available from https://www.gov.uk/government/statistical-data-sets/structure-of-the-agricultural-industry-in-england-and-the-uk-at-june

De Rensis, F.;Scaramuzzi, R.J. Heat stress and seasonal effects on reproduction in the dairy cow - A review. Theriogenology, 2003, 60(6), 1139–1151.

Dickin E and Wright D. 2008. The effects of winter waterlogging and summer drought on the growth and yield of winter wheat (Triticum aestivum L.). European Journal of Agronomy 28: 234-244.

Di Falco S, Chavas JP: Crop genetic diversity, farm productivity and the management of environmental risk in rainfed agriculture. Eur Rev Agr Econ. 2006, 33: 289-314. 10.1093/eurrag/jbl016.

Dunn N, Adlam J (2019). Establishing a resilient water supply. AHDB Facthseet 06/19

Dunn, R.J.H., Mead, N.E., Willett, K.M., Parker, D.E., 2014. Analysis of heat stress in {UK} dairy cattle and impact on milk yields. Environ. Res. Lett. 9 (6), 64006.

D. El Chami, J.W. Knox, A. Daccache, E.K. Weatherhead, (2015). The economics of irrigating wheat in a humid climate – A study in the East of England. Agricultural Systems, 133: 97-108

Entec (2006). An Investigation into Polytunnel Development in AONBs and National Parks. Report for The Countryside Agency. Available from http://www.polytunnelcontrol.org.uk/Polytunnels_Final_Report_Jan_06_with_Appendix_C_inserted.pdf

Evans N (2013). Strawberry fields forever? Conflict over neo-productivist Spanish polytunnel technology in British agriculture. Land Use Policy 35: 61-72

Foulkes M J., Sylvester-Bradley R & Scott R K. 2001. The ability of wheat cultivars to withstand drought in UK conditions: resource capture. Journal of Agricultural Science, Cambridge 137, 1–16.

Haro-Monteagudo, D, Knox JW, Holman IP. (2019). D-Risk: A decision-support webtool for improving drought risk management in irrigated agriculture. Computers And Electronics In Agriculture 162: 855-858

Haro-Monteagudo, D A. Daccache, J.W. Knox. (2017). Exploring the utility of drought and water scarcity indicators to assess climate risks to agricultural productivity in a humid climate. Hydrology Research, 49(2): 539-551. Doi 10.2166/nh.2017.010

Hess T, Knox J, Holman I, Sutcliffe C (2020). Resilience of primary food production to a changing climate: on-farm responses to water-related risks. Water 12, 2155 DOI:10.3390/w12082155

Hingley M, Lindgreen A and Casswell B. 2006. Supplier-Retailer Relationships in the UK Fresh Produce Supply Chain, Journal of International Food & Agribusiness Marketing, 18:1-2, 49-86, DOI: 10.1300/J047v18n01_04

Holman I, Knox J, Hess T, McEwen L, Salmoral G, Rey D, Hannaford J, Grove I, Thompson J, Quinn N (2019). Coping with drought and water scarcity: lessons for the agricultural sector. https://doi.org/10.17862/cranfield.rd.16810828.v2

Holman IP, Hess TM, Rey D, Knox JW (2021). A multi-level framework for adaptation to drought within temperate agriculture. Frontiers in Environmental Science Sci. 8:589871. doi: 10.3389/fenvs.2020.589871

Holman I.P. and Trawick P (2011). Developing adaptive capacity within groundwater abstraction management systems. Journal of Environmental Management, 92(6), 1542-1549 http://dx.doi.org/10.1016/j.jenvman.2011.01.008

Holman IP, Brown C, Janes V, Sandars D (2017). Can we be certain about future land use change in Europe? A multi-scenario, integrated-assessment analysis. Agricultural Systems 15: 126–135

Keay, C.A.; Jones, R.J.A.; Hannam, J.A.; Barrie, I.A. The implications of a changing climate on agricultural land classification in England and Wales. J. Agric. Sci. 2014, 152, 23–37

Kemp, D., Kenyon, R., Morris, S.T., 2010. The use of legume and herb forage species to create high performance pastures for sheep and cattle grazing systems. R Bras Zootec 39, 169–174.

Knox JW, Kay MG, Hess TM, Holman IP (2020). The challenges of developing an irrigation strategy for UK agriculture and horticulture in 2020: industry and research priorities. CAB Reviews 15: 050

Knox JW, Hess TM (2019). High level review of the Optimum Water Use methodology for agriculture following the 2018 drought in England. Cranfield University Technical Briefing Note for the Environment Agency.

Knox, J. and Hess, T. (2018) 'Advances in irrigation management and technology in potato cultivation: experiences from a humid climate'. In Wale S (ed.) Achieving sustainable cultivation of potatoes, Volume 2, pp. 69–88. doi: 10.19103/as.2017.0031.05.

Knox JW, Haro-Monteagudo, Hess TM, Morris J (2018). Identifying Trade-Offs and Reconciling Competing Demands for Water: Integrating Agriculture Into a Robust Decision-Making Framework. Earth's Future 6(10): 1457-1470

Knox, J.W., Daccache, A., Weatherhead, E.K., Groves, S., andHulin, A. (2013), 'Assessment of the impacts of climate changeand changes in land use on future water requirement andavailability for farming, and opportunities for adaptation(FFG1129): (Phase I) Final Report', Defra, London

Knox, J.; Morris, J.; Hess, T. (2010). Identifying future risks to UK agricultural crop production: Putting climate change in context. Outlook Agric. 39, 249–256

Knox, J.W.; Morris, J.; Weatherhead EK, Turner AP (2000). Mapping the financial benefits of sprinkler irrigation and potential financial impact of restrictions on abstraction: A case-study in Anglian Region. J. Environ. Manage. 58: 45-59

Klocke NL, Payero JO and Schneekloth JP. 2007. Long-term response of corn to limited irrigation and crop rotation. Transactions of the ASABE 50 (6): 2117-2124

Klocke NL, Currie RS, Kisekka I and Stone LR. 2014. Corn and grain sorghum response to limited irrigation, drought, and hail. Applied Engineering in Agriculture 30 (6): 915-924

Kuchenmeister, K; Kuchenmeister, F; Kayser, M; Wrage-Monnig, N; Isselstein, J (2013). Influence of drought stress on nutritive value of perennial forage legumes. Int. J. of Plant Production 7(4): 693-710

Lambert, R; Van der Veeren, B; Decamps, C; Cremer, S; De Toffoli, M; Javaux, M (2020). Forage production and drought: developing solutions for Wallonia. Fourrages 244: 31-376

Lea-Cox JD, Ross DS and Teffeau KM. 2001. A water and nutrient management process for container nursery and greenhouse production systems in Maryland. Journal of Environmental Horticulture 19: 230 – 236

Leathes W, Knox JW, Kay M, Trawick P, Rodriguez-Diaz JA, 2008. Developing UK farmers' institutional capacity to defend their water rights and effectively manage limited water resources. Irrig. Drain. 57: 322-331

Lee, J.M., Clark, A.J., Roche, J.R., 2013. Climate change effects and adaptation options for temperate pasture-based dairy farming systems: a review. Grass Forage Sci. 68 (4), 485–503.

Lee MA, Howard-Andrews V, Chester M (2019). Resistance of Multiple Diploid and Tetraploid Perennial Ryegrass (Lolium perenne L.) Varieties to Three Projected Drought Scenarios for the UK in 2080. Agronomy-Basel 9(3): 159. DOI 10.3390/agronomy9030159

Mackay I. Horwell A, Garner J, White J, McKee J and Philpott H. 2011. Reanalyses of the historical series of UK variety trails to quantify the contributions of genetic and environmental factors to trends and variability in yield over time. Theoretical Applied Genetics 122: 225-238. DOI 10.1007/s00122-010-1438-y

Marshall, A.H., Collins, R.P., Humphreys, M.W., Scullion, J., 2016. A new emphasis on root traits for perennial grass and legume varieties with environmental and ecological benefits. Food Energy Secur. 5 (1), 26–39.

Mechler, R, Hochrainer, S, Aaheim, A, Salen, H, Wreford, A (2010). Modelling economic impacts and adaptation to extreme events: Insights from European case studies. Mitig. Adapt. Strategies for Global Change 15(7): 737-762. DOI 10.1007/s11027-010-9249-7

Morris, J., Weatherhead, E. K., Mills, J., Dunderdale, J. A. L., Hess, T. M. and Gowing, D. J. G. et al. (1997). Spray Irrigation Cost Benefit Study. Final Report to the Environment Agency. Cranfield University, Bedford

NABIM. 2018. Wheat testing at mill intake. NABIM: London: https://www.ukflourmillers.org/wheattestingatmillintake

National Farmers Union (2019). Why potato growers need to be better prepared for drought. https://www.fwi.co.uk/arable/crop-management/irrigation/why-potato-growers-need-to-be-better-prepared-for-drought (accessed 06/03/23)

Neal, J.S., Fulkerson, W.J., Lawrie, R., Barchia, I.M., 2009. Difference in yield and persistence among perennial foliages used by the dairy industry under optimum and deficit irrigation. Crop Pasture Sci. 60, 1071–1087.

NFU (2015) Water availability in a changing climate: A survey of irrigated crop growers in the river Lark & Wissey catchments. National Farmers Union, UK. https://www.nfuonline.com/wissey-and-lark-water-use-report-may-15/

OECD. 2018. OECD Fruit and Vegetables Standards: https://www.oecd.org/agriculture/fruit-vegetables/

Palmer RC and Smit RP (2013). Soil structural degradation in SW England and its impact on surface-water runoff generation. Soil Use and Manage. 29: 567-575

69. Parsons D, Rey D, Tanguy T, Holman IP (2019). Regional variations in the link between drought indices and reported agricultural impacts of drought. Agricultural Systems 173: 119-129

Rey, D., Holman, I.P, Sutcliffe C, Hess T (2023). Synergies and trade-offs in drought resilience within a multi-level UK food supply chain. Regional Environ. Change 23: 55. DOI: 10.1007/s10113-023-02046-x

Rey, D. Water option contracts for reducing water supply risks. Outlook Agric 2015, 44, 5–9.

Rey, D.; Pérez-Blanco, C.D.; Escriva-Bou, A.; Girard, C.; Veldkamp, T.I.E. (2019). Role of economic instruments in water allocation reform: Lessons from Europe. Int. J. Water Resour. Dev. , 35, 206–239

Rey D., Holman, I.P., Daccache, A., Morris, J., Weatherhead, E.K., and Knox, J.W. (2016). Modelling and mapping the economic value of supplemental irrigation in a humid climate. Agr Water Manage. 173: 13-22

Rey, D., Holman, I.P. and Knox, J.W. (2017). Developing drought resilience in irrigated agriculture in the face of increasing water scarcity. Reg. Environ. Change 17(5): 1527-1540

Rey Vicario D, Holman I, Knox J. (2021) Towards the operationalization of water sharing for irrigation in England, Project Report, July 2021, Cranfield University. http://dspace.lib.cranfield.ac.uk/handle/1826/17290

Rial-Lovera, K.; Davies, W.P.; Cannon, N.D. Implications of climate change predictions for UK cropping and prospects for possible mitigation: A review of challenges and potential responses. J. Sci. Food Agric. 2017, 97, 17–32

Riley WD, Potter ECE, Biggs J, Collins AL et al. (2018). Small Water Bodies in Great Britain and Ireland: Ecosystem function, human-generated degradation, and options for restorative action. Sci Total Environ. 645: 1598-1616.

Rio M, Rey D, Prudhomme C, Holman IP (2018). Evaluation of changing surface water abstraction reliability for supplemental irrigation under climate change. Agric Water Manage. 206: 200-208

Roy MG, Hough MN and Starr JR. 1978. Some agricultural effects of the drought of 1975-76 in the United Kingdom. Weather 33 (2): London: Meteorological Society. 64-74

Salmoral G, Rey D, Rudd A, de Margon P, Holman IP (2019). A probabilistic risk assessment of the national economic impacts of regulatory drought management on irrigated agriculture. Earth’s Future 7(2): 178-196 DOI: 10.1029/2018EF001092

Salmoral G, Ababio B, Holman I (2020). Drought impacts, coping responses and adaptation in the UK outdoor livestock sector: insights to increase drought resilience. Land 9(6): 202

Sheaffer, C.C.; Peterson, P.R.; Hall, M.H.; Stordahl, J.B. Drought effects on yield and quality of perennial grasses in the north central United States. J Production Agriculture, 1992, 5(4): 556-561

Sutcliffe, C., Knox, J. and Hess, T. (2021) ‘Managing irrigation under pressure: How supply chain demands and environmental objectives drive imbalance in agricultural resilience to water shortages’, Agricultural Water Management, 243, p. 106484. doi: 10.1016/j.agwat.2020.106484.

Talbot M. 1984. Yield variability of crop varieties in the UK. Journal of Agricultural Science 102: 315-321

Wallace EE, Chappell NA (2020). A statistical comparison of spatio-temporal surface moisture patterns beneath a semi-natural grassland and permanent pasture: From drought to saturation. Hydrol. Proc 34(13): 3000-3020. DOI 10.1002/hyp.13774

Ward D and McHague K. 2007. Water requirements of livestock. Factsheet 07-023. Ontario Ministry of Agriculture, Food and Rural Affairs.

Weatherhead, K., Knox, J., Hess, T., & Daccache, A. (2015). Exploring irrigation futures: developments in demand forecasting. Outlook on Agriculture, 44(2), 119–126.

Weatherhead, E.K., Knox, J.W., Twite, C.L. and Morris, J. (2002) Optimum Use of Water for Industry and Agriculture, Phase III Technical Report. Part B - Agricultural component. EA Water Resources R&D Project W6-056/TR

Weatherhead, E.K., and Knox, J.W. (1999), 'Predicting and mapping the future demand for irrigation water in Englandand Wales', Agricultural Water Management, Vol43, pp 203–218

Whaley, L., and Weatherhead, E. K. (2015a). Power-sharing in the English lowlands? The political economy of farmer participation and cooperation in water governance. Water Altern. 8 (1), 820–843.

Whaley, L., and E. K. Weatherhead. 2015b. Competition, conflict, and compromise: three discourses used by irrigators in England and their implications for the co-management of water resources. Water Alternatives 8(1):800-819.

Whaley, L., and E. K. Weatherhead. 2015c. Using the politicized institutional analysis and development framework to analyze (adaptive) comanagement: farming and water resources in England. Ecology and Society 20(3):43. http://dx.doi.org/10.5751/ES-07769-200343

WMO and GWP, 2016: Handbook of drought indicators and indices: Integrated Drought Management Programme (IDMP), Integrated Drought Management Tools and Guidelines Series 2. WMO/TD-1173, 52 pp., https://www.droughtmanagement.info/find/guidelines-tools/handbook-drought-indicators-and-indices/

Wreford A, Adger WN. (2010). Adaptation in agriculture: historic effects of heat waves and droughts on UK agriculture. Int J. Agric. Sustain. 8(4): 278-289. doi:10.3763/ijas.2010.0482

Wreford A, Topp CFE (2020). Impacts of climate change on livestock and possible adaptations: A case study of the United Kingdom. Agric Systems 178: 102737. DOI 10.1016/j.agsy.2019.102737

J: Vegetation drought responses, wildfires and their impact on water quality

Jill Thompson, France Gerard, Douglas Kelley, Rebecca Oliver, Amy Pickard, Maliko Tanguy, Maud van Soest.

UK Centre for Hydrology & Ecology.

Overview

Droughts in the UK are expected to increase in frequency and severity over a greater area as climate change progresses. Drought causes changes to flowering and fruiting phenology, low productivity, increased herbivore damage, disease and ultimately plant death. As a result drought impacts carbon storage and cycling, biodiversity, our food supply and wildfire occurrence. In addition to the direct impact of drought through low water availability, droughts are commonly associated with higher temperatures, increased wind and vegetation drying that increases the chance of vegetation fires.

Terrestrial plants access water from soils that have specific capacities to retain soil moisture where water availability depends upon soil mineralogy and chemistry, soil structure and organic matter content. Water availability also depends upon plant roots, and microorganisms such as bacteria and fungi. In addition to rooting depth and architecture, bacteria may secrete compounds e.g. osmolytes that impact water flow from the soil into roots, while mychorrhizal fungi affect root physiology, and increase effective root surface area and thus the soil volume accessed for water and nutrients.

Plants vary in drought tolerance within and among species, reflecting their functional traits, (e.g. leaf shape, size and thickness). Plant responses and adaptation to drought includes, drought escape (e.g. changing phenology and life cycle), drought avoidance (e.g. control of stomata and shedding leaves), drought tolerance (e.g. resisting dehydration through physiology), and drought recovery where plants protect structures and are able to recover post drought. Drought response of different vegetation types depends upon characteristics of individual species their interactions and the environment. There are few examples of direct measurements of plant drought resilience and most information has been based upon observations of species and their environments.

The vegetation type, and thus the vegetation and soil water holding capacity, the timing and length of drought conditions and the resultant fuel load from dead and dry vegetation determines the potential for wildfires. Drought associated high temperatures and wind, and the site location and time of year contribute to devastating fires. As fires have not been a natural feature (e.g. over evolutionary time scales) few if any UK species have evolved to resist fire such as think bark on trees or potential regrow after fire. However, some plants have protected growing points under the soil and may regrow. Vegetation recovery after wildfires can take many years and depends upon the extent of vegetation destruction and impact on the soil, availability of seed sources and management. Before vegetation recovery the possibility of soil loss through erosion by wind and water is a major concern, and water runoff pollutes waterways. Vegetation fires release much carbon and particulate pollution into the atmosphere and those that ignite heavily organic soils such as peat can burn long term. The sporadic nature of drought in the UK and the danger has made assessment of drought wildfires difficult. Attempts to model the potential for fires and their spread have received increasing attention but are currently not able to adequately predict them.

The potential use of Earth Observation (EO) for monitoring vegetation drought has been steadily increasing. Drought impacts on plant functions such as photosynthesis, chlorophyll fluorescence, leaf water and temperature, can be seen through changes in their spectral images, as well as anatomical changes and colour changes due to plant mortality. Some 60 EO based indicators can now be used to monitor drought. Agricultural drought and soil water can be assessed through microwave backscatter. Methods assessing plant biomass can look at the effects of drought while estimates of plant water stress are more useful for predicting future drought. Integrated indexes such as Vegetation Condition Indexes are increasingly useful and machine learning and Bayesian techniques are developing research areas. The index used may depend upon vegetation type.

The land surface component of Earth System Models (ESMs) simulates exchange of carbon, water and energy between the land surface and atmosphere and allows exploration of changing environmental conditions over large spatial scales to investigate historical and to predict future droughts, through changing gradients in soil climate and other environmental conditions. Despite the high uncertainty in ESM predications of carbon cycle dynamics because of the complex biological processes involved ESMs formed the main tool for UN Intergovernmental Panel on Climate change projections for reports to policy makers.

Vegetation drought and wildfire research needs

We need more information on all aspects of drought monitoring, predicting and impact on vegetation, and the models and earth observation needed to assess droughts; including specifically:

1. Species specific information on anatomical and physiological plant responses to drought and water stress impact on factors that can be assessed through earth observation – such as changes in in their hyperspectral images.

2. Species specific differences and variation within species in drought tolerance and resistance, especially for UK species. Standard methods to assess plant drought independently of their environment. Also the genetic basis of such variation so that if possible ecotypes, varieties and species can be selected for drought prone areas and to select for climate change in the future.

3. nteractions between plant species, and the biotic environment including other plant species, microbes and fungi as well as the abiotic soil environment and the impact of soil nutrients and biochemistry on drought resistance. Also the impact of species communities in different vegetation types and the impact of facilitation and competition on vegetation drought resistance and resilience.

4. Potential for species and vegetation types in the UK to suffer wildfires and recover after fires. The impact of wildfires on atmospheric pollution and water courses, and soil structure and function, and potential for post fire soil recovery. Controlled experiments are required to reduce the variability and discover mechanisms.

5. Climatic, vegetation and abiotic conditions that lead to drought and increased potential for wildfires.

6. The data collected through 1-5 will contribute to earth observations that should increase the potential to develop predictive models of drought and fire and their impact.

1. Summary scope of this essay

1. Introduction. Definitions and scope of this review – Summary of the impact of drought on vegetation in a range of habitats, the potential for wildfires and their impact on water quality.

2. Impact of drought on plant species and vegetation in different habitats forest, grassland and heathland

3. Potential for drought mitigation through vegetation structure, and species selection and acclimatization.

4. Impact of drought and related wildfires on water quality (e.g. pH, suspended sediments, organic matter and nitrogen) in at risk habitats.

5. Vegetation drought detection through Earth Observation monitoring and changes in evapotranspiration signals.

6. Modelling and forecasting drought induced wildfire events.

7. Summary of knowledge gaps and research needs to further our understanding of drought impacts on vegetation, and wildfires.

1.1. Introduction and scope: Drought, soil, plants, vegetation and wildfires

Drought for plants is impractical to define (Lloyd-Hughes 2014) because it depends upon the attributes of the subjects of the drought, their evolutionary history, the impact of the climate regime and the environment in which the drought occurs. Drought is, therefore, a general qualitative term, defined by Sheffield and Wood (2011 Chpt 1), as a "deficit of water relative to normal conditions". Sheffield and Wood (2011 chpt 2) also noted that "Drought is a silent and pervasive disaster that creeps up over weeks and months, often without warning" and that it "can cover extensive areas for many years with devastating impacts".

Despite the difficulty in defining the parameters that cause or describe a drought for any location, particular species and plant community, we recognize that a drought in vegetation is occurring or has happened when we see plants wilting, turning brown and dying, crops failing, soils may be cracked and dusty and there may be vegetation fires.

The impact of drought on individual plants and vegetation types depends upon the local environment and the usual climatic conditions that the vegetation is adapted to, how long reduced precipitation continues and associated weather effects such as temperature and wind, and the distribution of precipitation that ends the drought. It is difficult to determine at which point low precipitation will become a drought, and how intense the drought will be in any particular place. This is because the occurrence of a drought also depends upon the rate of drought development, the time in the growing season that a drought occurs, the species and plant community affected and their usual growing seasons, plant phenology and species adaptability, such as their ability to change their form and their physiology (Basu et al., 2012)

Drought occurrence, frequency and intensity is projected to increase in the UK and other parts of the world as a result of global warming (see other essay). Although droughts, as a result of extreme cold, can impact vegetation (Olsen et al., 2018), these habitats are not common in the UK and are likely to diminish in area with climate change. So are not dealt with here. We will also not deal in detail with the cellular (Takahashi et al., 2020), biochemical (Wani 2018) and genetic details (eg Shinwari 2020, Bashier et al., 2021), of plant drought tolerance, as these are a major study in their own right and outside the scope of this essay.

The details of UK drought history and predicted future scenarios are dealt with in another essay. We will focus here on droughts caused by lack of water and potentially associated with high temperatures, give an overview of ecological drought impacts on plant species and vegetation types and their potential to acclimatize to drought. Drought may lead to vegetation wildfires, which have major impacts during the fire itself, and for post-drought vegetation recovery. We will consider the impact of drought and wildfire on natural/semi natural vegetation (agricultural drought is covered by another essay), soils and watercourses. Finally, we will consider the potential for assessing drought using Earth Observation and remote sensing, and the use of ecosystem models.

1.2. Drought and the Soil environment

Before considering the plants themselves, to understand droughts' effect on vegetation, it is important to know how plants interact with soils from which they extract the water via roots and associated fungi (see below). A detailed review of drought and soil is covered in another essay, but we summarize important plant soil features here to provide context. Each soil has a specific capability to retain water as soil moisture, and the soil water availability for plants depends upon soil properties such as parent material and soil mineralogy, soil texture (proportion of sand, silt and clay), soil structure including pore size, pore networks, and organic matter content. Soils form aggregates of soil material and the size of pores and surface area of aggregates they form determines the extent to which the water can interact (through adhesion) with the soil and move through it. Soils with large aggregates and large pores may hold a greater volume of water, and release it readily, which can lead to low water tables inaccessible to roots, and high rates of evaporation from the soil. Small soil pores hold water tightly through interactions with soil material surfaces, which limits water movement and makes it harder for plants to extract water. A change in soil volume in response to changing water content can cause clay soils to shrink and swell causing soil cracking, which may increase the rate of water loss to deeper soil layers and cause physical damage to plant roots and expose roots to a dry atmosphere. Cracking also changes water infiltration patterns and rate of water loss through evaporation from deeper soil layers. Soil organic matter increases water-holding capacity, but drought reduces the strength of organic matter binding of soil into aggregate particles. Soil aggregates are unstable and may collapse during drought, resulting in smaller soil pores, lower water holding capacity, greater water adhesion, and soil surface sealing. Surface sealing occurs when dry layers of fine soil particles infill the pores and reduce the soil water infiltration capacity, so precipitation runs away rather than infiltrates into the soil, and plant roots are less able to penetrate soils to extract water. The soil structure and composition together determine the soil water holding capacity and availability of water to the plants.

Without precipitation, there is no evaporative cooling, so dry soils will reach higher temperatures (Turner et al., 2021) that may directly kill plant roots and disrupt fungal connections. Dry soils are more vulnerable to erosion from wind, and when rainfall returns erosion from surface water flow. Water arriving at the ground surface as precipitation will infiltrate or run off. Some moisture may be directly evaporated back into the atmosphere and never enter the rooting zone where it is available to plants. The water that infiltrates moves down into the soil and redistributes throughout the soil profile. Water may move laterally through the soil or move beyond the plant rooting zone into a saturated zone providing recharge to ground water.

In addition to the physical attributes of soil that determine water availability to plants, as noted above, the soil also provides nutrients to plants. Nutrients availability is affected by soil moisture, as low soil moisture reduces the rate of decomposition of soil organic matter to release nutrients, reduces nutrient movement through the soil profile, and the direct uptake of the nutrients into the plants, either directly through the plant roots (Sondergaard et al., 2004), or through associated fungal networks. Prolonged low soil moisture may also cause reduced availability of nitrogen and macronutrients, through water limited slow decomposition of plant litter and reduced activity of microorganisms. During prolonged drought, loose litter may accumulate on top of the soil surface as it fails to decompose and acts as a ready fuel source for fire ignition.

1.3. Other weather related impacts on drought

In addition to the lack of precipitation that directly causes a drought, other aspects of weather are important. During periods of low rainfall there is often less cloud cover that allows more intense sunlight to reach the land. Intense sunlight causes high temperatures that increase the rate of evaporation from the soil surface, so the impact of low rainfall on soil moisture will occur more rapidly and soil will dry to deeper depths. As well as causing higher soil temperatures that increase evaporation from the soil, heat in the atmosphere increases air movements and wind, which in turn increase evaporation from the soil and the evapotranspiration rate from the plant. High temperatures, and intense sunlight from clear skies, can cause direct effects on plant physiology through increased leaf temperatures, which denatures leaf proteins, and sunlight may bleach leaf chlorophyll and thus reduce photosynthesis.

1.4. Direct effect of drought on plants

The effects of drought can be devastating on plant growth and survival. At the whole-plant level, the effect of drought is evident as decreased CO2 assimilation and growth, leading to eventual mortality if the drought conditions are severe enough. Under drought stress, plants can experience carbon starvation, hydraulic failure, and increased vulnerability to disease, pests and fire (Choat et al., 2018). Plant browning and shedding of leaves may act as a defence mechanism to survive drought periods (Bogati and Walczak 2022). In areas where short term droughts often occur, an increase in root carbon content and root relative to shoot biomass enables the plant to “search” for more water.

The main effects of drought on plants is a reduction in the water stream from the roots to the leaves and water movement throughout the plant. Once inside the plant water is required to move nutrients to the parts of the plant where they are needed or through the xylem vessels and from cell to cell, and in turn water is required to move the products of photosynthesis and protein production throughout the plant (Sondergaard et al., 2004). Plant parts (mainly leaves) have stomata that are actively controlled pores through which carbon dioxide enters the plant for photosynthesis to produce carbohydrates for energy and growth, and from which water exits the plant in a process known as evapotranspiration. Evapotranspiration creates a ‘water stream’ movement from the roots to the stomata via xylem vessels and throughout the plant tissues. When soil moisture is not readily available and to prevent high rates of water loss from the leaves, the stomata close. As leaf water deficits increase in response to a decrease in either atmospheric humidity or soil water potential, carbon fixation through photosynthesis is reduced through enhanced diffusive resistances within the leaf (closure of stomata and/or decline of mesophyll and chloroplast conductance), and drought-induced impairments of metabolic processes (Flexas et al., 2006, Lawlor and Cornic, 2002, Yordanov et al., 2000). Plants exhibit either isohydric or anisohydric strategies. An isodydric strategy of water regulation where plants close stomata to avoid drought-may induce hydraulic failure that runs the risk of depleting plant carbon reserves if the drought is long. Hydraulic failure results from reduced soil water supply coupled with high evaporative demand causing xylem to cavitate or collapse, preventing water flow through the plant and desiccating plant tissues. Plants that follow anisohydric behaviour typically suffer from hydraulic failure, maintaining higher rates of stomatal conductance during drought stress to enable carbon uptake. Although considered more drought-tolerant, anisohydric strategies predispose plants to hydraulic failure (and development of air embolisms in xylem vessels) because they operate with narrower hydraulic safety margins during drought, and if the drought is sufficiently intense plants risk running out of water before running out of carbon (McDowell et al., 2008). The risks of hydraulic failure increase when water availability declines across the plants’ rooting zones or vapour pressure deficit increases at their leaf surfaces. Closed stomata, to prevent excessive water loss through transpiration and hydraulic failure, reduces the uptake of carbon dioxide for photosynthesis may result in carbon starvation. This stomatal closure and reduction of internal water movement also prevents nutrients and other plant produced products such as carbohydrates and proteins travelling to where they are needed throughout the plant.

1.5. Plant Growth Promoting Bacteria (PGPB) and Fungal Mycorrhizae

Bacteria are the important microorganisms of plant root rhizospheres that affect plant physiological and biochemical activities. PGPB have important role in plant growth and development, they provide plant growth promoting substances to their host plants and in turn get protection and food in the form mutualistic approach. These bacteria secrete various compounds in the form of osmolytes, antioxidant, phytohormones etc. that enhance the root osmotic potential under drought stress. (Ullah et al., 2019a, b). In addition, they also regulate the plant growth under drought stress, either by direct (enhanced production of phytohormones) or indirect through increasing availability of nutrients. The soil microbial community can improve the drought tolerance of plants (Kannenberg and Philips 2017, Kour and Yadav 2022, Rubin et al., 2017). Plant growth-promoting rhizobacteria (PGPR) are responsible for mitigating drought stress in dry environments (Niu et al., 2018) and inoculating PGPR into crop plants can increase their drought tolerance (Gontia et al., 2016). In experiments with cereals, microorganisms were found to play a vital role in reducing the adverse effects of drought stress, thereby improving plant productivity (Khan et al., 2020) . The oxidative damage in the plants grown under different environmental stresses can be reduced through the presence of microorganisms and enabling the cereals to cope with drought conditions.

Mycorrhizae are fungi that form an association with host plants and can increase drought tolerance through physiological and molecular processes. Fungal hyphae external to the roots increase the effective root area exposure to soil and build an extremely branched mycelia that can connect plants together in a mycorrhizal network (Simard et al., 2015). The mycelium can absorb water from a greater volume and deeper soil, which is then transported to cortical plant tissues and through plant tissues. Arbuscular mycorrhizal fungi (AMF) are one of the most important soil microbes and function as symbionts with roots of plants (Brundrett and Tedersoo 2018) they enhance growth of the host plant via promoting water uptake and nutrients to control abiotic stresses, such as drought stress (Bowles et al., 2018).

Chandrasekaran (2022) found that during droughts plant root physiology may change to alleviate drought stress by enhancing AMF colonization (Wu et al., 2017), and increasing nutrient (N, P, K) uptake. Soil K is a driver of changes in root morphology and may assist in the alleviation of plant drought stress (Xu et al., 2021). The soil community can also produce a legacy effect of drought on plant performance of species as tested by Buchenau et al., (2022) on the grass species Lolium perenne, Bromus hordeaceus and Alopecurus pratensis, and the forb species Centaurea jacea, Diplotaxis tenuifolia and Prunella vulgaris. The interplay between plants, microbes and fauna in the soil determines the magnitude of drought stress the ecosystem experiences (see review by Malik et al., 2022).

1.6. Other impacts of drought on plants

Drought does not always kill plants, but can affect them in other ways at different points in their life cycle, and at different times of the year. The temperature during a drought is often higher than usual and its effects depend upon the time of year and the life stage of the plant. Spring and autumn droughts are often cooler and, therefore, less damaging than mid-summer droughts, where lack of precipitation combined with hotter temperatures makes drought more intense and damaging. Spring droughts may affect flowering, pollination and thus future fruit set and seed production. Naturally dispersed or sown seed may be killed or suffer reduce seed germination, and subsequent seedling death. Drought in the middle of the growing season will affect overall plant growth and productivity, and fruit and seed development that will reduce future species survival chances. These are particularly damaging for agricultural crops. Late season drought may affect bud development for the next year in perennial plants and crops.

Stress caused by low soil moisture and high temperature also affects plant susceptibility to disease, pests and fire (Choat et al., 2018). Water is required to produce secondary protective compounds in response to pathogen and pest attack, using soil nutrients, and carbohydrates and proteins produced in plant leaves that are then transported using water to the region needed. Drought stress closes stomata, reduces photosynthesis and protein production, and slows water transport through the plant (Vasquez-Gonzalas et al., 2022) This potentially prevents plant defences protecting them from attack by herbivores and pathogens

1.7. Plant responses and adaptions to drought.

Vegetation responses to drought are affected by the timing of drought (duration, severity and magnitude), in addition to when drought occurs in the phenological cycle. For example, a drought event that occurs in the height of the growing season will result in greater growth reductions and/or higher plant mortality than drought during a less active growth period (Drewniak and Gonzalez-Meler, 2017). A severe drought with a long duration will impact plant response and survival differently than short, frequent droughts.

Therefore, the time a plant has been exposed to drought cycles is an important driver of trait changes that increase drought tolerance (Alves et al., 2020, Drewniak and Gonzalez-Meler, 2017). Acclimation to drought, that is the acquisition of the ability to tolerate drought, includes both genotypic and phenotypic changes, with water stress shown to induce the expression of genes that are associated with adaptive responses of stressed plants (Yordanov et al., 2000). Water stress experienced over longer time scales, particularly in long-lived plants such as trees, can lead to acclimation, rendering plants less susceptible to the negative impacts of drought (e.g. xylem cavitation) (Zhou et al., 2016). At the ecosystem level, this coupling of the carbon and water cycles affects gross primary production (GPP) rates and evapotranspiration in response to water stress (Jaideep et al., 2020).

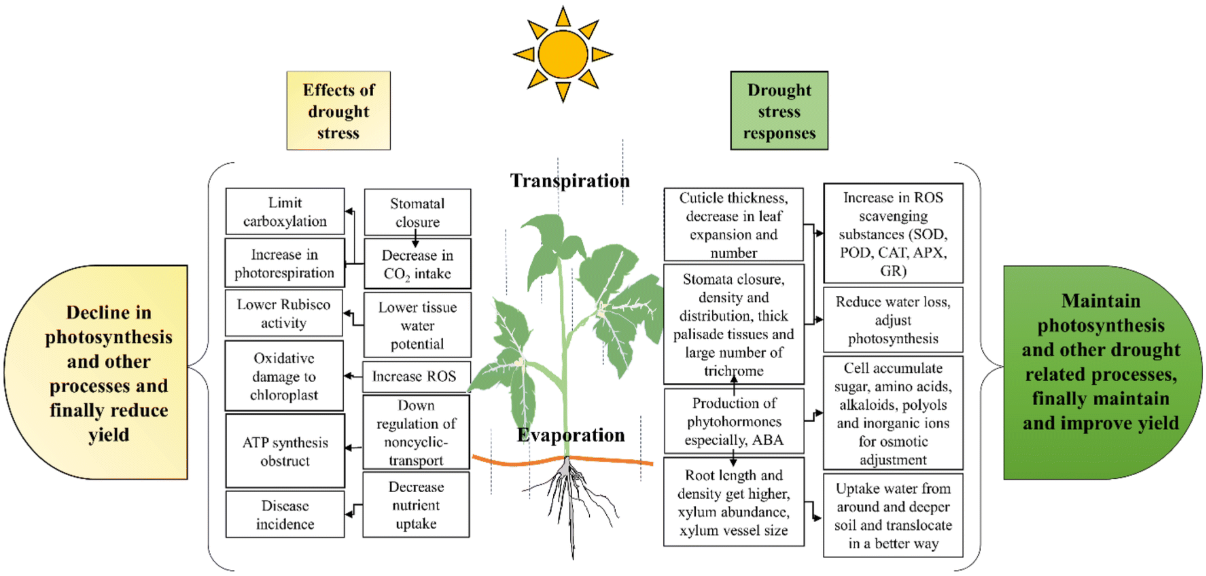


Figure: Effects of drought stress and morpho-physiological responses in plants, from: [Drought Tolerance Strategies in Plants: A Mechanistic Approach](https://doi.org/10.1007/s00344-020-10174-5) Ilyas, M., Nisar, M., Khan, N. et al., Drought Tolerance Strategies in Plants: A Mechanistic Approach. J Plant Growth Regul 40, 926–944 (2021). https://doi.org/10.1007/s00344-020-10174-5 Reactive oxygen species (ROS)

Plants have evolved different water management strategies and functional traits to control the balance between stomata being open for carbon dioxide to enter and for evapotranspiration to occur to maintain internal plant physiology, and closing stomata to prevent too much water loss. Species vary in the number of stomata and location of stomata and the degree to which they can control them. Some plants can finely control their leaf water, while others are more adapted to variable leaf water and are able to keep their stomata open for longer and to draw water from drier soils (Maherali et al., 2004). In stressed conditions, plant adaptations and acclimation that allow stomata to stay open and maintain transpiration in dry conditions could lead to surface cooling when compared to when drought causes complete cessation of transpiration. Studies on urban trees demonstrate the cooling effect of transpiration from vegetated surfaces (Winbourne et al., 2020).

On exposure to long-term drought, plant acclimation can include adjustments at different levels including leaf physiology, anatomy, morphology, chemical composition, xylem hydraulics, growth and/or carbon partitioning (Zhou et al., 2016). These changes in allometry (i.e. increase in water absorbing roots and decrease in transpiring foliage) occur to optimise hydraulic conductance and maintain the water transport capacity of the plant (Choat et al., 2012, Magnani et al., 2002, Martin-StPaul et al., 2013). Decreasing leaf area is often observed to reduce water use, maintain leaf water potentials, and therefore buffer leaves from effects of water limitation and desiccation (Limousin et al., 2010, Martin-StPaul et al., 2013). Changes in xylem anatomical features, such as the production of more cavitation resistant xylem (e.g. adapting conduit diameter, wall thickness or pit membrane features) can also increase plant tolerance to drought (Hacke et al., 2001, Lens et al., 2011).

Plant responses and adaptions to drought have been divided into four types 1) Drought escape, 2) Drought avoidance, 3) Drought recovery and 4) Drought tolerance. Plant species may have a combination of these types; each type has evolved a wide range of variations (Fang and Xiong 2015, Basu et al., 2016). 1) Drought escape includes plants changing their life cycle to mitigate drought stress. If water is not available at the time for seed germination, flowering or fruiting and the plant has phenotypic plasticity, they may be able to adapt and change their phenology and thus mitigate the effects of drought (Kramer 1995). 2) Drought avoidance protects the plants through behaviours such as control of stomata to reduce evapotranspiration, dropping leaves, and increasing depth and density of roots to increase access to available water resources. 3) Drought recovery is the ability of plants to restart growth and reproduction after exposure to drought stress. Some species drop their leaves decrease the plant structures they need to maintain, and reduce transpiration and rate of water loss before low water potential causes plant mortality. After the drought has finished the plant regrows leaves or roots (Manavalan et al., 2009) and the plant continues to survive. Loss of leaves also reduces the tension on the water column in the xylem vessels and this reduces the chance of embolisms that could permanently block the xylem vessels and kill the plant. 4) Drought tolerance enables plants to resist dehydration through physiological activities such as production of osmoprotectants (Luo 2010). Drought resistance can include all four of these mechanisms.

The impact of drought depends upon the species and their specific functional morphological and structural traits. Plant height, xylem structures and hydraulic traits (Barros et al., 2015) affect plant internal water movement, leaf area and number of stomata per unit area, affect evapotranspiration rates, while rooting depth and root fungal associations affect access to water from the soil.

Other functional morphological traits that impact drought responses include root form and rooting depth and pattern, which are frequently species specific, and also depend upon surrounding soil characteristics (sections above, e.g. Thomas 2000) and other abiotic factors. Combinations of traits are species specific and some of these traits vary more among individuals of the same species, than among different species particular in tropical forests. Plant genetics physiology and biochemistry are also important but are not dealt with here (see e,g, Ilyas et al., 2021). In addition, leaf size and shape, cuticle thickness (Read et al., 2003), and distribution of cell types within plants all have their individual effects. Two tree species Picea abies and Fagus sylvatica that commonly grow in the same regions represent diverse hydrologic anatomy and physiology, which affects their drought responses. For example, Picea abies (spruce) is an evergreen gymnosperm with tracheids, few stem parenchyma cells, needles, and a mostly shallow-rooting system, while Fagus sylvatica (beech) is a deciduous angiosperm with xylem vessel elements, a higher proportion of xylem parenchyma, broadleaves, and a heartrooting system, with coarse roots spreading horizontally and vertically from the rootstock, with its fine root distribution below that of spruce (Grams et al., 2021).

The development of some traits often depends upon the environment in with a plant is growing, so there may be a legacy effect. Some plants in an environment with frequent, short-term, low-intensity droughts may suffer less during a drought than plants that have been established in wet areas, which are then subjected to unusual drought conditions. Hasibeder et al., (2015) found that short term summer drought altered the carbon allocation to grass roots, which enabled plants to withstand subsequent droughts more easily. A phenology-growth trait complex may evolve in response to differences in precipitation. For example, populations of Abies alba from areas characterized by generally low levels of precipitation have evolved to start the growing season early, grow slowly, and have a high water use efficiency (Csilléry et al., 2020).

In the UK droughts tend to co-occur with high temperatures (as noted above). In response to increasing temperatures experienced within the growing season, numerous studies show that plants can adjust their photosynthetic rates to maintain higher carbon assimilation rates. Thermal acclimation of photosynthesis in response to rising temperatures typically increases the optimum temperature for photosynthesis towards the new growth temperature, which increases or maintains the photosynthetic rate respective to the growth temperature (Yamori et al., 2014). However, numerous studies show plants can acclimate to their environment, in that the optimal temperature for photosynthesis is adjusted to growth conditions experienced over the timescale of days to weeks(Dusenge et al., 2020, Slot et al., 2021, Way and Yamori, 2014). Acclimation of photosynthetic capacity to drought has been shown in Eucalyptus species (Zhou et al., 2016).

1.8. Drought and Species Response

As noted above, during drought spells, plant systems actively maintain physiological water balance by (i) increasing root water uptake from the soil, (ii) reducing water loss by closing stomata, and (iii) adjusting osmotic processes within tissues (Rodrigues et al., 2019; Gupta et al., 2020; Seleiman et al., 2021). Plants in their usual habitats adapt to their environment through variety of means ranging from transient responses to low soil moisture, to major survival mechanisms of escape by early flowering if seasonal rainfall is absent (Basu et al., 2016).

The sheer diversity of plant species grown across climatic regions including extreme dry conditions demonstrates that plants have evolved to endure degrees of drought stress with an array of morphological, physiological, and biochemical adaptations (Bohnert et al., 1995). As noted above it is the difference between the usual climate conditions, particularly water availability that the plant usually experiences, and the relative reduction in water availability and its interaction with other environmental factors, which results in a “drought” experience for particular plant species in a particular location.

Species of similar functional type regarded as components of a homogeneous group (sclerophylls) that all showed a high degree of sclerophylly (hard leaves that are tough and leathery) and lived in the same environment, adopted completely different ways to avoid drought (Read et al., 2003). Mediterranean species Olea oleaster behaved as a ‘drought-tolerating’ species. Drought was ‘avoided’ by Ceratonia siliqua by a ‘water-spending’ strategy and by Laurus nobilis by a ‘water-saving’ strategy combined with the capability of recovering even minimal water losses by drastically dropping leaf water potential (Lo Gullo 1988). Different species exhibit different vulnerabilities to drought impacts (Blackman et al., 2016, Locatelli et al., 2021) and develop different ‘coping strategies’ which can be grouped in two categories (Jump et al., 2017): drought-avoidance and drought tolerance

Even within species, plants display very different sensitivities to drought. For example, three genotypes of the perennial bioenergy grass Miscanthus showed contrasting responses to drought. One displayed stomatal regulation of water loss by reducing leaf conductance and photosynthesis to retain green leaf area, even under severe water shortage. The other two varieties lost leaf area under drought by senescence (Clifton-Brown et al., 2002). Differences in whole-plant water use efficiency were not detected in the three varieties, suggesting that the ‘best’ strategy for drought survival would depend on drought timing, frequency and magnitude (Clifton-Brown and Lewandowski, 2000). Miscanthus sacchariflorus and M. x giganteus would be better suited when droughts are normally short, however if droughts are prolonged, M. sinensis may survive better as it is able to maintain leaf area and continue growth after the drought period has passed. Clones of Poplar balsamifera seedlings showed different responses to drought, although all were anisohydric they showed different degrees of regulation of leaf water potential, shedding leaves stem growth and water use efficiency (Ryan 2011).

Forest research (https://www.forestresearch.gov.uk/climate-change/adaptation-measures/tree-species-diversity/) recognizes that drought sensitivity varies among tree species. For example, beech, birch and sycamore species are more sensitive than hornbeam or native oak species. Species such as Douglas fir and western hemlock are more tolerant of drought and show lower susceptibility to stem cracking than species such as Sitka spruce. An experiment on beech and spruce demonstrated species differences between spruce and beech. After a drought, one week after precipitation returned the pre-dawn leaf water potentials of drought-stressed trees had recovered, but stem diameter growth did not re-establish in the same growing season, and remained reduced by 33% in beech and 69% in spruce compared with controls over several years (Grams et al., 2021).

Blackman (et al., 2016) proposed an index of desiccation leading to tree mortality. However, reports on species drought attributes are often based upon observations of the environmental conditions in which they normally grow, and their responses during past droughts. While these observations are valuable to determine impact of climate change, for example the phenological observations of Betula pubescens (birch), Fagus sylvatica (beech) and Quercus robur (oak) in The Netherlands (Kramer et al., 1995). Results from seedlings may not reflect what happens to mature trees as tree breeders have long known that seed or seedling traits are often poor predictors of adult traits in field conditions (e.g., Resende et al., 2012), with some exceptions, e.g., wood traits (Gaspar et al., 2008) or seed size in pines (Zas and Sampedro (2015). Measures of adult growth traits in-situ may also not be helpful to predict drought responses when plants are affected by management practices and interspecific competition (Kunstler et al., 2011).

UK forest research has a site assessment tool for selecting where to plant trees, but it does not yet include species information for dealing with droughts. (https://www.forestresearch.gov.uk/tools-and-resources/fthr/ecological-site-classification/).

There appears to be are few direct tests of the drought responses of UK native species (such as those for Acer tree seedlings, Khalil et al., 1992), the strategies they have evolved, their genetic variation or the plasticity in their drought tolerance (Madouh et al., 2022). Some information can be found from other countries, such as in The Netherlands where several grass species that are also found in the UK including Lolium perenne, Bromus hordeaceus and Alopecurus pratensis (Buchenau et al., 2022) have been tested for drought. Species lists for drought tolerant garden plants such as those by Kew Gardens https://www.kew.org/read-and-watch/drought-loving-plants-garden and the Royal Horticultural society https://www.rhsplants.co.uk/drought-resistant-plants are mostly not native.

There have been experimental tests for the drought tolerance of crop plants, because of their economic and food value. Seliman (2021) lists drought tolerance mechanisms for wheat, soya, sunflower, cotton, canola, maize and rice, and see the review by Daryanto et al., (2020) on a variety of important crops. There is also research into breeding crop plants for drought tolerance (Varshney et al., 2021), but agriculture and droughts are the subject of another essay.

1.9. Drought and vegetation type

Different vegetation types have different species combinations and develop in different environmental conditions that define them. The combination of species and their functional traits, species interactions and interactions with the abiotic environment determine how vegetation will respond to drought.

Vegetation types with low species and functional diversity may resist drought if the individual species are drought tolerant. For example agricultural species that have been developed specifically to be drought tolerant may suffer less than a diverse species community of semi natural vegetation that has not evolved drought strategies. Similarly low diversity communities in desert landscapes where individual species have evolved drought tolerance, may be less damaged by drought when compared to high species diversity communities with non drought resistant species. However, within a particular location, high diversity species communities are generally more drought resistance that monocultures, or low diversity vegetation. Drought resistant high diversity plant communities may be more drought resilient because some vegetation cover may be maintained through some deep rooted species having easier access to water from deeper depths. At the same time, those with more responsive stomata may reduce water loss and not wilt nor drop their leaves. Although some species may be lost from the community, if some plants persist, the vegetation is ready to respond to renewed rainfall, to absorb water reduce erosion and runoff and hold the soil together. The impact of drought on individual plant species, and how species interactions in plant communities respond in different vegetation types, in different abiotic environments, with different plant soil feedbacks requires much detailed research (Veresoglou et al., 2022).

1.9.1. Drought in forests

The potential for forests to be damaged by drought on UK is increasing, and there is evidence that areas previously considered unlikely to suffer drought will be impacted. Drought risks to forest are affected by the forests location and although summers are projected to become drier across most of the UK their impact will be site and forest specific such as predicted for east central and southern Scotland. Large scale tree mortality not yet observed in Scotland but severe damage was observed in Scotland the UK and Scotland after 2003 and 2018 droughts (Locatelli et al., 2021). In Scotland, direct effects of severe droughts are likely to be felt primarily on forest productivity. Ovenden et al., (2021) report a substantial reduction in radial stem growth of Scots pine in Scotland for a period of approximately 5 years after a drought event. The largest reductions in productivity from drought on forests are likely in southern England, where temperatures will also be warmest. Site attributes such as soil type are also important, and forests on shallow, light-textured and freely draining soils that hold less water will be more affected by reduced precipitation than those on deep, heavier clay soils. However, trees growing on deep soils, but poorly drained so that they are waterlogged in winter may have shallow rooting zones, and therefore may be at risk if upper soil layers dry out in

summer, even if there is water in deeper layers
https://www.forestresearch.gov.uk/climate-change/risks/drought/

In addition to drought that leads to tree mortality, drought can lead to reduced tree growth, stem cracking, crown dieback, susceptibility to biotic attacks, or pathogens (Desprez-Loustau et al., 2006) and increased risk of windthrow and wildfire. Few studies have investigated the interactions between drought and other disturbances, with fire and pests found to be the predominant co-stressors (Jactel et al., 2012).

New tree planting, older, and veteran trees are more likely to be most affected by drought. As the effects of drought differ with species and tree age, there will likely be changes to stand age structure and species composition over time and with repeated droughts. Newly planted trees will be particularly vulnerable to droughts, especially those on exposed open sites as are smaller, more fragmented woodlands as these will be more likely to are more likely to dry out than extensive forest areas when sunlight temperature and wind exposure are reduced. Forests are adaptable ecosystems, however, constraints on natural processes can impact forest recovery after a drought and the degree of drought disturbance is linked to the precipitation history before the drought occurs and forest condition at the time of the drought.
https://www.forestresearch.gov.uk/climate-change/risks/drought/

Species composition is significant in determining large-scale responses to drought, given the variation of the ability to acclimate to water stress observed within and between species. While severe drought may trigger tree mortality within a population, some trees will be less vulnerable to dry conditions than others and survive (DeSoto et al., 2020). It is suggested that the resilience of forest stands and trees to drought is enhanced by increasing diversity in functional species traits (Ammer, 2019) and diversity in hydraulic strategies (Anderegg et al., 2018). Species that have the capacity to acclimate to drought conditions will therefore be beneficial in maintaining the long-term persistence of the global land carbon sink

As species differ in their vulnerability to drought forests of Scots pine, Douglas fir, and Sitka spruce in the UK are expected to be most impacted, however provenance trials have shown large variability in drought vulnerability among provenances, which are as great as between species. Modelling climate and tree growth through tree rings (dendrochronology) may provide information on drought tolerance of some species and different species provenances.
(https://www.climatexchange.org.uk/research/projects/drought-risk-in-scottish-forests/).

Although trees often cope with soil moisture deficiencies in one year, damage will occur if the soil moisture store is not replenished. Forest recovery from drought depends upon the timing and intensity of the drought, and whether there have been repeated droughts that had a cumulative effect. Predicting the resilience and recovery of trees and forests to drought is hindered by the unknown predisposition of trees entering the drought event.

This includes the cumulative effects of previous water deficits in combination with other environmental factors (e.g. pest outbreaks) that influence survival, mortality and recovery of stress from drought (e.g. Anderegg et al., 2007, Choat et al., 2018)

Results from the first assessment of the large species and provenance trials of the REINFORCE European consortium across a long Atlantic seaboard transect confirm the particular vulnerability of young trees (4-year old) to drought. Correia (2018) report that mortality in the broadleaves was driven by the size of the difference between precipitation in the native and non-native ranges, while conifer survival was affected by elevated accumulated temperature. For broadleaved species, this suggests that precipitation levels in the region of the source seed material, must be carefully compared, with those of the intended planting location. For conifers it is important to ensure that the differences in temperature and the length of the growing season between the source and target locations are kept to a minimum. This study identified a strong negative correlation between tree height and a drought index derived from accumulated temperature and precipitation. Conifers were more heavily impacted than broadleaves, despite the latter being more susceptible to unfavourable soil properties than conifers.

Observations on mixed versus monoculture forest stands reveal that Douglas-fir in mixed stands exhibit a significant improved growing performance compared to pure stands. European beech seems to react indifferently concerning its performance in mixture compared to pure stands (Thrum et al., 2016). Differences in drought stress resistance and growth recovery time mainly arose between the species. Douglas-fir showed a significantly lower resistance and required more time to reach its initial growth level again compared to European beech. After drought in a species mixture there was a trend for shorter recovery time for Douglas-fir growth and longer recovery time for European beech after a drought period (Thrum et al., 2016).

A global forest study by Jump et al., (2017) showed the importance of looking at forests rather than individual species, when environmental favourability declines, increases in water and temperature stress that are protracted, rapid, or both, drive a gradient of tree structural responses that can modify forest self-thinning relationships. They expect forests to become increasingly structurally mismatched to water availability during more stressful episodes. Forest research (https://www.forestresearch.gov.uk/climate-change/risks/drought/) noted that drought risk needs to be assessed and adaptation measures implemented to help manage these risks. Research and ongoing monitoring can help better understand the risk and impacts of drought on different tree species, provenances and sites.

Drought effects result from the complex interplay between climate extremes, many different components of forest ecosystems and other biotic and abiotic disturbances. All of these elements will be affected by climate change in ways that cannot be confidently projected, which makes predicting the interactions among them very difficult. (https://www.climatexchange.org.uk/research/projects/drought-risk-in-scottish-forests/)

Silvicultural strategies such as underplanting (planting seedlings of shade-tolerant species such as western hemlock, Douglas fir, and western red cedar, in an established stand) increases forest diversity the additional structural complexity provided can also have positive effects on drought resilience, such as by reducing air temperatures and wind speed, and increasing humidity in a multi-storey stand structure (Stokes et al., 2020). Mixed species stands also provides drought resilience as different species can withstand different degrees, extent and timing of droughts. Recent evidence suggests that forests with high structural and species diversity are more resilient and better adapted to climate change impacts (Kirby et al., 2009, Ray et al., 2015).

Other forest management practices such as thinning forests potentially allows for the development of larger root systems that may increase future resilience to drought [38]. However, thinning may also promote the growth of ground vegetation and thus competition for soil water; more ground vegetation growth also increases the risk of wildfire spread. In some cases, thinning the overstorey can increase air temperature and reduce humidity at canopy level, which may increase tree water loss (Sohn et al., 2016). A selection of forest drought risk mitigation strategies can be found here https://www.climatexchange.org.uk/research/projects/drought-risk-in-scottish-forests/

Resources and current projects concerning forest drought:

Project PRAFOR Probabilistic drought Risk Analysis for FORested landscapes An ongoing at UK forest research and European institutions to develop probabilistic methods, environmental risk assessments and process based models for forests, climate predictions and drought impacts. Dealing mainly with conifer species including scots pine. (https://www.forestresearch.gov.uk/research/prafor-probabilistic-drought-risk-analysis-for-forested-landscapes/).

There is a recently launched hub for forests and drought in UK https://www.forestresearch.gov.uk/climate-change/risks/drought/

Forest research has a decision support tool to choose the best forest species for a site is available for a range of species and uses average climate conditions but it does not yet have capacity for determining the impact of possible extreme events such as drought. https://www.forestresearch.gov.uk/tools-and-resources/fthr/ecological-site-classification/

1.9.2. Drought in Grasslands

The degree to which plant species diversity in grasslands and herbaceous mixtures may promote resilience to drought has yielded inconsistent results with some mixtures showing diversity increased, decreased or neutral impacts on biomass production depending on the species community, level of productivity and management (Vogel et al., 2012). More productive fertile grassland with high relative growth rate species may be less able to withstand drought (Grime 2000). Vogel et al., (2000) hypothesized that high species richness has a positive effect on productivity but this might result in a lower resistance to climate change. Grassland studies experiments in California showed that reduced precipitation had little effect on productivity, but the species community shifted as the competition among species changed (Van Dyke et al., 2022). Some of the inconsistency of grassland drought responses may be difficulties in considering the changes in species richness with productivity measurements. Species mixes where most species have similar functional traits may be similarly affected by drought, and the trend may be negative between species richness and productivity despite high species richness. Hussain et al., (2022) found that biodiversity positively affected resistance to drought but mixed effects when considering resilience. Vogel (2012) also noted the impact of grassland management and timing of cutting on their grassland drought resistance. With few exceptions (e g Grime 2008 in the UK, and Hossain 2022 in Germany) most grassland drought experiments have been short lived and the variability in results difficult to tease out. Rainfall reduction experiments in wet years may not reduce water sufficiently to cause drought stress and show a difference between reduced precipitation and control treatments (Ayling et al., 2021). Houssain (2022) emphasized that plant species diversity stabilize grassland ecosystem productivity and increase resistance to climate change. Improvements to grassland drought experiments have been suggested by Matos et al., (2020), including a focus on the exploring the transitions between wet and dry phases and climate extremes when assessing vegetation resilience to climate change.

The ability for grassland species to withstand drought depends upon the soil, its microbes and its drought legacy (Xi et al., 2022). A study by XI (2022) in planted mesocosms reported that: (i) drought decreases bacterial and fungal richness and modified the relationships between plant species richness and microbial groups; (ii) drought soil legacy increased net biodiversity effects, but net plant species richness was unaffected; and (iii) linkages between plant species richness and complementarity/selection effects depended on past and subsequent drought.

Ongoing international experiments on grassland and drought:

– Drought-Net: A global network merging observations, experiments, and modelling to forecast terrestrial ecosystem sensitivity to drought. Established in 2016. Principal Investigator(s):Kate Wilkins, Colorado State University; Osvaldo Sala; Peter Wilfahrt, University of Bayreuth; Laureano Gherardi; Melinda Smith: https://ui.adsabs.harvard.edu/abs/2016AGUFM.B11J..06S/abstract

– New Drought synthesis project https://lternet.edu/working-groups/a-global-synthesis-of-multi-year-drought-effects-on-terrestrial-ecosystems/ Published January 16, 2023

1.9.3. Drought in heathlands

Calluna vulgaris (heather) and Deschampsia flexuosa are dominant heathland plants in the UK. Kongstad (2012) reported the high resilience in heathland plants to drought and high temperatures. After a 3 year experiment by Konstad et al., (2012) they found that drought reduced the growth of the two dominant species Deschampsia flexuosa and Calluna vulgaris (Brandbjerg, Denmark). However, both species recovered quickly after rewetting and the drought applied had no significant effect on annual aboveground biomass production. This heathland plant species community also did not show any significant responses to the imposed climate changes and they concluded that these two heathland species, on a short time scale, will be relatively resistant to potential changes in climatic conditions. However Kondstad et al., (2012) suggested that the ability of D. flexuosa and C. vulgaris to recover from severe drought may change in the future as periodic drought events permanently change growth conditions over time. However, Seaton et al., (2022) suggested that in heathlands longer term and more intense (up to some level) than normal droughts may have a reduced effect as the ecosystem is able to adapt as the drought changed the distribution of Calluna vulgaris roots. There were also changes to soil bacteria and fungal communities in upland heathland organic soil, which may affect the future drought resistance of these heathland communities. Drought increased the dominance of fungi over bacteria in temperate heath (Haugwitz 2014).

1.10. Knowledge gaps, species, vegetation and drought

Drought and temperature response of UK native plant species, their structural and physiological adaptability, within species variation and responses to different environments is required, and a comprehensive database with this information established. Potentially invasive species should be included in order to predict outcomes under climate change scenarios.

A public searchable database of drought tolerant UK native plants would be useful such as that for Oklahoma plants https://www.edmondok.gov/1494/Drought-Tolerant-Plant-Database

Interactions among plants and the impact of species competition and facilitation during drought, and in different environments, are required in order to understand and predict the future.

Long term and large scale experiments on natural and semi-natural and plant communities are required, to look for resistance and resilience of established vegetation.

Forest-scale experiments are urgently needed to improve our understanding of trees' responses to extreme drought events and subsequent recovery under field conditions (Grams et al., 2021). To address questions on the extent that forests will be affected by a changing climate, experiments should consider a range of forest ages and species arrangements, from saplings/seedlings to mature forest stands (e.g., Lola da Costa et al., 2010, Pangle et al., 2012).

Efforts to determine tree drought tolerance by modelling climate and tree growth through tree rings (dendrochronology). Tree rings show some evidence that Douglas fir shows drought resistance. Investigations in a stand of mature Douglas fir stands identified that high wood density and high late-wood to early-wood ratio were associated with lower likelihood of drought-induced mortality in affected stands (Martinez-Meier 2008). (https://www.climatexchange.org.uk/research/projects/drought-risk-in-scottish-forests/).

To allow forest managers to make informed decisions about implementing drought-reducing silvicultural practices, the forecasts of drought-related risks to growth and productivity of Scottish forests estimated by Petr et al., (2014), will require higher spatial resolutions, and a better representation of the fundamental interactions between forest stands, soils, and climate.

More investigation is needed to understand how high temperature stress interacts with drought, particularly the impact on photosynthetic capacity and biochemical acclimation.

2. Drought and wildfires, impact on soils and vegetation.

2.1. Introduction

"Wildfire" are "an unusual or extraordinary free-burning vegetation fire which may be started maliciously, accidentally, or through natural means, which negatively influences social, economic, or environmental values" (UNEP et al., 2022). Whether caused by humans or nature, fires can become wildfires when they burn out of control. These contrast with landscape fires - fires that local people and ecosystems are more accustomed to and are critical to many ecosystems' healthy functioning and essential cultural and land management tools.

The impacts of wildfires are experienced over various timescales. The immediate consequences include loss of life, wildlife, habitat and crops, and destruction of property and infrastructure. The longer-term impacts include environmental damage that may last for many years. Drought, leaf drop and vegetation death provides dry fuel that together with heat and an ignition source may lead to vegetation fires. Different vegetation types will burn in different ways depending on fuel distribution, local weather, and abiotic conditions. The intensity of the burn, its temperature and length of burn time of the fire have profound effects on the soil, vegetation, watercourses and their subsequent recovery. The increase in wildfires around the world is mostly via an increase in fuel availability where more dry vegetation is available to burn, and the fires season is lengthening through increased number of days when metrological conditions are conducive to fire (Jones et al., 2020, 2022).

Drought has a major influence on fire occurrences that can be categorised as: (1) direct effects on fire weather through drought, higher temperatures, which, when coupled with higher winds, make fuels drier and easier to burn (Littell et al., 2016); (2) indirect effects resulting from changes in the nature and availability of biomass/fuel; and; (3) direct and indirect changes in the frequency and location of natural and human-caused ignitions via changes in dry lightning profiles, and changes in demographics and human behaviour (Dale et al., 2001; Krawchuk and Moritz 2011; McKenzie and Littell 2017; Restaino and Safford 2018).

The impacts of drought and wildfire on soil and wider ecosystem functions are of increasing interest. The processes and responses vary from very small to large ecosystem scales. More multidisciplinary ecosystem-based studies are required because of the complexity of the different possible effects and reasons for potential changes. Long-term monitoring experiments are rare and, if and how ecosystems recover after drought or fire is relatively unknown. To understand recovery from fire, more data and knowledge about soil processes before and during drought and fire events is required. The importance of timing of drought and wildfires events is acknowledged. However, information on the disturbance to nutrient availability is required as this will affect plant growth and survival at different times of year. Newly developed models like SheFire (Brady et al., 2022) will help explore how wildfires and soil heating influence physical, chemical and biochemical responses. Ultimately, soil moisture deficits can lead to a shift in plant species and physiological changes in vegetation more vulnerable to fires (Mansoor et al., 2022).

2.2. Impacts of wildfire on soils

The greatest proportion of heat released in fire is transferred to the air while the heat, which reaches the ground surface, is transferred into the soil profile through a combination of radiation, conduction, convection, mass transport, vaporization, and condensation (Stavi, 2019). Interestingly, there is no correlation between fuel load, fire intensity, and fire damage to soil (Stoof et al., 2013). Burn severity is a term used to describe the loss of organic matter in the soil and degree of soil burning (Parsons et al., 2010).

The direct and indirect of fire on soil depends on the soil temperature reached and the duration of heating, direct effects range from a reduction in soil organic matter, soil porosity, saturated conductivity, soil water retention and infiltration capacity to increases in dry bulk density and development of fire-induced soil water repellence (Stoof et al., 2015). Mataix-Solera et al., (2011) showed that fire could cause both soil aggregation, when strong aggregates could form due to the recrystallisation of certain minerals, and disaggregation when organic matter was destroyed. Jain et al., (2012) presented a classification of five post-fire soil environments ranging from unburned, to absence of surface organic matter, to aid the interpretations of fire effects research. The combined effects of fire on soil physical and hydrological changes that change soil properties governing water flow and soil stability increase the risk of overland flow generation and soil vulnerability to erosion (Stoof et al., 2015). Indirect fire effects on soils are caused by soil exposure after the vegetation is removed, increasing the vulnerability to erosion (Stoof et al., 2015).

2.3. Impacts of wildfire on vegetation

Wildfire is a major environmental process because it removes the vegetative cover, leaving bare soils more prone to soil erosion. Although wild fires have not been a major occurrence in past decades in the UK. The predicted drought frequency and intensity increase is likely to increase wildfires (e.g., the Saddle Moor fires in 2018). While many plant species from fire-prone ecosystems have evolved and are adapted to specific fire regimes corresponding to historical conditions, changes from low fire occurrence or shifts from the normal regime of fire adapted ecosystems may have severe impacts on vegetation recovery, and long-term species persistence (Tangney et al., 2022). Fires outside of the historical fire season may lead to decreased post-fire recruitment, particularly in obligate seeding species, but a post fire increase in abundance of re-sprouting species is possible.

Research on how fire season influences fundamental plant life-cycle stages is surprisingly limited (Miller et al., 2017) and often focuses on the local scale. Fire seasonality is a component of the fire regime and shapes plant responses to fire. How plants respond to fire season variation is linked to seasonal phenological cycles of growth, dormancy, flowering, and reproduction, meaning that when fire occurs during vulnerable phenological stages, post-fire population recovery responses can be negatively impacted.

The frequency and extent of wildfires is influenced by the intersection of climate, weather, ignitions and landscape fuels (Bradstock et al., 2010). Both the short-term characteristics of fire events and long-term, interval-related elements of fire regimes can shape species distributions. The characteristics of individual fire events filter plant functional types in the short-term, leading to local mortality or recruitment failures, while the overall fire regime can impart selective pressures on species and traits within ecosystems over generations (Keith et al., 2007). As wildfires fall outside the range of disturbance normally experienced by local ecosystems, they can overwhelm plants adaptations to fire.

In areas regularly affected by fire, many trees have evolved thick bark to protect xylem from flames and heat. As wildfires become more intense, hotter flames that burn for longer may burn through the bark, to increase tree mortality either during the fire or soon afterwards. Imported species such as giant redwoods do have distinctly thick and insulating bark. Other adaptations to fire include the ability to re-sprout from buried roots and shoots that are protected from the heat by the soil. In the UK native tree species are unlikely to have evolved thick bark protection, and few plants will have evolved other fire adaptations as historically fires have not been frequent enough to create sufficient selection pressure for fire adaptations. Combined, drought and wildfire causes more stress to plants, causing increased plant mortality (Hato et al., 2021; Machado-Silva et al., 2021).

For UK forests there are two periods of high wildfire risk: late winter-spring when there is dead-dry ground vegetation present (e.g., grass, bracken), overnight-frosts, dry periods with low daytime relative humidity; and summer, with hot and dry weather, including heatwaves and droughts. Wildfires during the summer are usually more intense and more damaging. The changing climate is likely to increase the risk of wildfires in both the late winter-spring and summer periods, and may extend the high-risk season into the autumn. Compared with warmer and dryer Mediterranean countries there have been relatively few forest fires but they do occur especially where grassland or heathland is close to woodlands or open rides within forests provide flammable material. After pest and disease outbreaks kill trees that are standing or where fallen trees are left to decay the fire risk increases because of the extra fuel loads.

There are three main types of forest wildfires: surface fires, ground fires and crown fires. The most common is the surface fire, where the understory plants such as heather, grass, bracken and gorse along forests edges and the understory in open forest areas create fire fuel. There can be substantial growth of these fuels along roads and rides, before canopy closure, and after woods have been thinned. Surface fires can burn fiercely, spread fast with long flames and at high fire intensity. Ground fires consume peat and soil organic matter and threaten carbon stores. Smouldering peat fires are hard to extinguish and can re-kindle frequently. Tree crown fires occur less frequently, during hot and dry summers, but are the most dangerous. They spread from surface fires, with ladder fuels such as tall shrubs, low tree canopies, and standing deadwood. Trees in poor health, and leaning windblown trees, increase the risk of canopy fires. There are a variety of methods to prevent forest fires such as fire breaks (see https://www.forestresearch.gov.uk/climate-change/risks/wildfire/ accessed 25 Jan 2023)

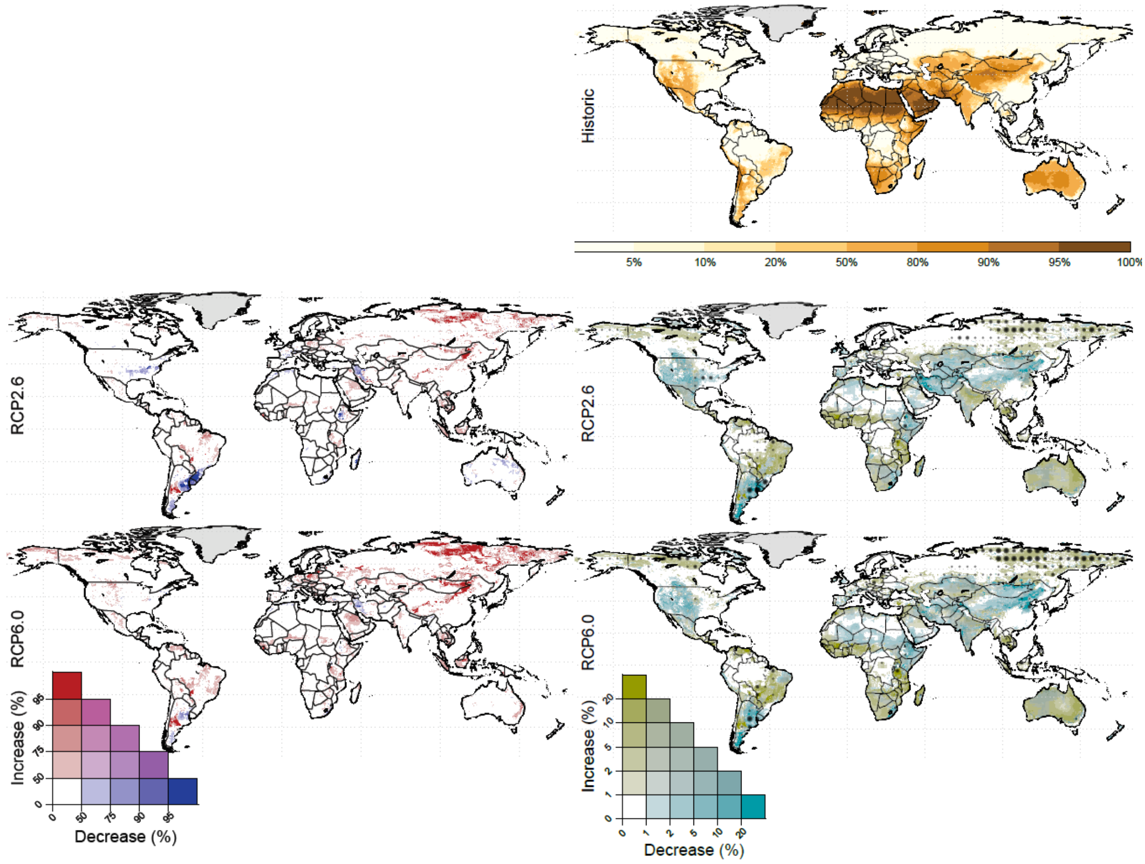
As with drought, soil-plant interactions are affected by changed soil conditions post fire. Some plants increase the allocation of biomass to roots to compensate for nitrogen limitation after fire, thereby changing the root-to-shoot ratio and leading to an increase in soil respiration (Stavi, 2019). On the other hand, restoration of vegetation can be relatively fast when rhizome buds of perennial herbaceous plants remain unharmed (Bond, 2001).

2.4. Future wildfires and wildfire drivers

With projections of more frequent droughts comes increases in wildfires (Figure above), along with a corresponding increase in burnt fallen and standing trees (UNEP et al., 2022. These devastating wildfires will likely become more severe and frequent, with

(UNEP et al., 2022) estimating a worldwide increase of 8-14% in the next 10 years and 31-57% by 2100.

Several factors influence the level of danger of a particular fire, including the environmental conditions such as the weather, available fuel and topography affecting the severity of the fire, human factors, landscape modification and resources for suppression (Kelley et al., 2019; Yeo et al., 2015). Climate change affects the weather that a fire needs to burn if ignited and suitable climate conditions for fuel (Flannigan et al., 2016). Whilst the wildfire has increased, the burnt area globally has decreased, mainly due to land fragmentation and fuel clearance for agriculture and fire suppression efforts in tropical savannas, which see extensive landscape fires (Knorr et al., 2016; Andela et al., 2017; Kelley et al., 2019).



Left: future changes in wildfire occurrence compared by 2090-2100 vs 2010-2020 for (top) RCP2.6 and (bottom) RCP6.0 future emissions scenarios. Right: Change in fuel dryness (right). The black dots indicate areas of a significant shift dryness. Reproduced from (UNEP et al., 2022)

Factors influencing fires vary for different regions worldwide, including both natural processes and anthropogenic behaviour. For example, both the western USA and Australia have a long history of fire across their ecosystems, partly driven by their weather and climate conditions, whereas weather and climate has had less influence on fires across Brazil and the Amazon (Alencar et al., 2015; Kelley et al., 2021; da Silva Junior et al., 2020).

2.5. Drought drivers and wildfire changes

There has not been any comprehensive evaluation of the drivers of wildfire globally, due to the relatively short observational record and global fire models inability to reproduce extreme fire events (Hantson et al., 2016; Hantson et al., 2020). However, we can infer areas where drought conditions are more likely to driver wildlife activity from sensitivity of burnt area to fuel moisture relative to other controls (Figure below; Kelley et al., 2019). High burning rates in tropical forests are more likely to occur due to low fuel moisture. The same is true in boreal forests, though here high fire activity is also limited by available ignitions. However, in fuel limited systems such as tropical and Mediterranean savannahs and grasslands, temperate woodland, and arid systems such as shrub and desert, low fire activity coincides with drier conditions because of the reduced fuel production. In large parts of Africa with annual average rainfall less that 800 mm, for example drought conditions are associated with less burnt area due to a decline in grass biomass and productivity (Alvarado et al., 2020).

Over the past two decades, droughts and heatwaves have become more frequent and severe (Geirinhas et al., 2018; Panisset et al., 2018) they also make it more difficult to suppress fire and increase the potential for outbreaks to become wildfires that burn unchecked for extended periods (Adams et al., 2020).

2.6. Models for wildfires spread

Climate change influences on fire are generally categorised as (1) direct effects of climate change on fire weather conditions such as drought, high temperatures, winds, and their seasonality; (2) indirect effects on fire through vegetation condition as climate change alters the structure, abundance, and energetics of biomass/fuel; and (3) changes in ignition potential due to shifting spatiotemporal patterns of lightning and alterations in human behaviour in response to factors such as climate policy and environmental management (Figure below) (Dale et al., 2001; Krawchuk and Moritz 2011; McKenzie and Littell 2017; Restaino and Safford 2018).

The impact of these complex interacting processes on fire regimes are not well understood and depend on whether factors act synergistically or antagonistically (Balch et al., 2009; Prentice et al., 2011). One option is to improve representation of fire in models, the challenges of which are covered in detail in (Hantson et al., 2016). The increase in observational data on fire properties such as burnt area, radiative power and fire size, and the development of statistical technique has led to the development of empirical assessments of fire drivers and impacts that can help inform model parameterisation (Haas et al., 2022) or link directly to fire processes (Forkel et al., 2017;

Kelley et al., 2019, 2021)]. However, many of these approaches have limited utility due to lack of vital process representation and feedbacks. Large scale model development is slow, and many approaches have been developed but it will take some time to develop models to the point where they can help us understand and predict fires spread.

Recent work within the western US using statistical models demonstrated that winter and spring climate conditions are critical for predicting summer burned area (Abolafia-Rosenzweig et al., 2022). Further work suggests that extreme weather conditions will be the major driving force for fires, surpassing our ability to implement effective management. The complexity of exploring fire behaviour and effects is related to scale, heterogeneity and feedbacks. There is a range of complexity across models from direct connections to fire weather to including interactions between climate and live and dead fuels (FireMIP; Rabin et al., 2017). The addition of processes (fuels, crown fires, within canopy climate, extinguish) increases both complexity and uncertainty, and is often limited by data availability for benchmarking.

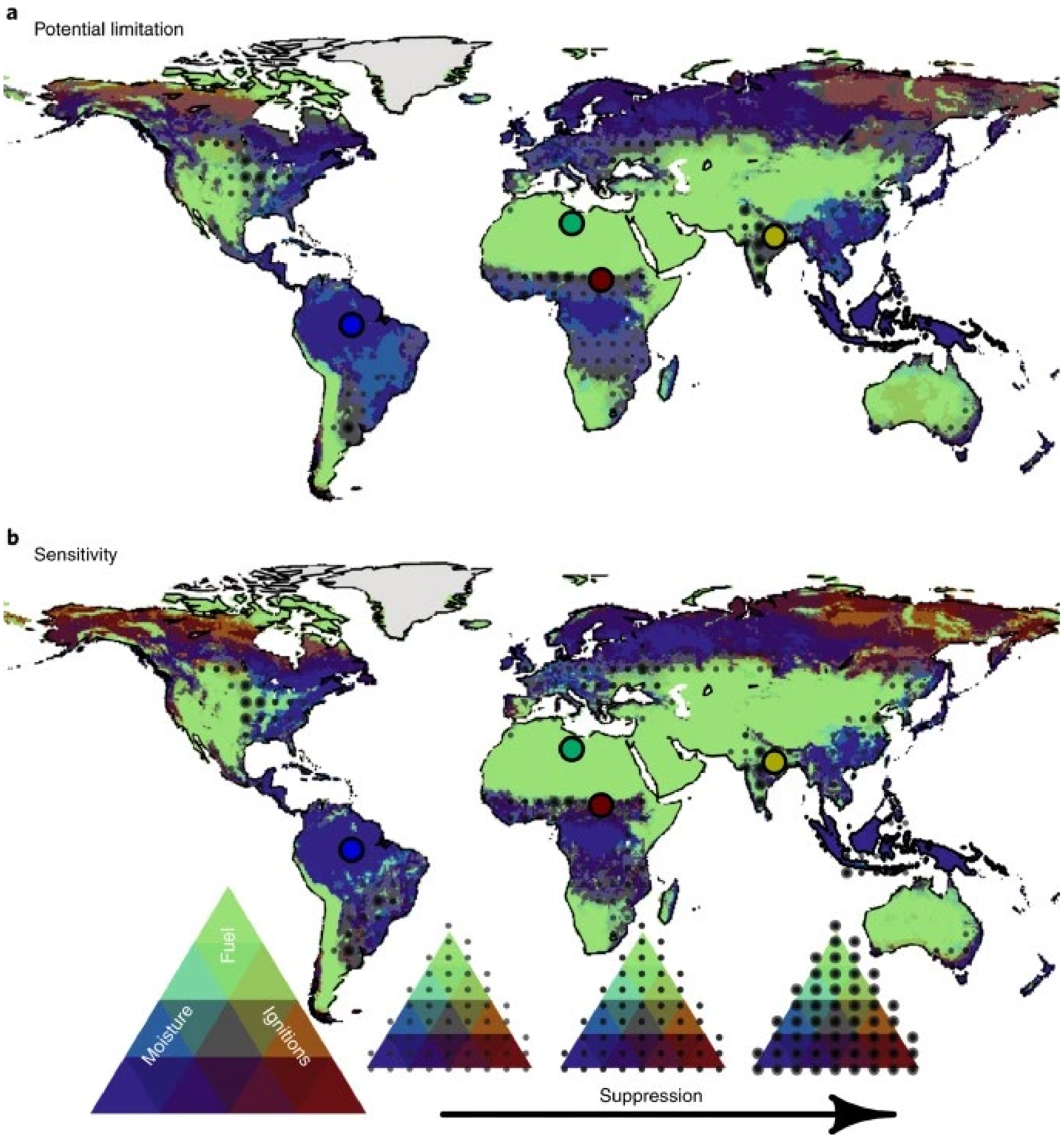


Figure: Areas where variations in the annual average burnt area are sensitive to: fuel (green); moisture (blue); ignitions (red); suppression (stippled); fuel and moisture (cyan); fuel and ignitions (brown); moisture and ignitions (magenta); fuel, moisture and ignitions (grey). Moisture areas are the areas most sensitive to drought. Reproduced from (Kelley et al., 2019).

2.7. Fire modelling research gaps and needs

Make progress on linking climate, ecological, and fire management research - identified as the biggest bottleneck for developing effective and future-proof fire management plans (UNEP 2022).

Reduce the uncertainty and biases in global models that are currently not able to represent the stochastic nature of fires.

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The impact of complex interacting processes on fire regimes are not well understood and depend on whether factors act synergistically or antagonistically (Balch et al., 2009; Prentice et al., 2011).

Need to develop more appropriate metrics used in fire model comparisons, to design data-model comparisons that help characterize models and identify appropriate use.

Fire is inherently dynamic with processes that vary across scales from flame to airshed to define a fire event (Finney et al 2015, Clements et al 2015). Improve the translation of the fire event scale to global application involves abstractions that obscures site-level heterogeneity known to influence fire behaviour, such as variability in topography, wind, fuel, and belowground conditions.

A central finding of the Fire Model Intercomparison project (FIREMIP) was that simple representations of anthropogenic impacts on fire – based on readily available data such as population density or GDP – are a substantial shortcoming in current global fire models. Including 1) the absence of a systematic empirical basis from which to derive improved representations of people in global models and 2) the lack of appropriate modelling frameworks through which to capture and project anthropogenic fire impacts globally.

3. Drought induced wildfire and the impact on water quality in the UK

3.1. Introduction

A detailed overview of the impact of drought on water quality is given in an accompanying essay, and here we focus on the effects of drought-induced wildfire on water quality. Wildfire affect vegetation and soils in many ways, and the creation of ash and changes in soil physical and biochemical properties provide a critical link between water infiltration, runoff, and erosion that deposits material into watercourses, with subsequent effects on water quality Many studies have been carried out in the USA (Paul et al., 2022), but few in the UK. Given the increasing occurrence of wildfire events in the UK, particularly over the years since 2019, there is a need to understand the likely water quality outcomes of these extreme climatic events.

The amount and chemical properties of ash produced during wildfires depends upon the vegetation destroyed and the fire intensity (Bodi et al., 2014). Ash may fill soil pores to reduce water infiltration into soil, and may be directly washed into watercourses. There have been few studies on the impact of wildfires in the UK because of their relative infrequency in the past, and the dynamic nature of wildfire events renders it difficult to establish water quality monitoring using a Before-After-Control-Impact (B-A-C-I) experimental design. More often, wildfire studies will use a control-impact assessment, or a before-after assessment, and Evans et al., (2017) remains the only study at the UK-scale to use this design. This is because the affected water body, Blue Lough, has been studied as part of UKCEH's UK Upland Waters Monitoring Network, where high quality data on sensitive freshwater ecosystems has been consistently collected since 1988. It is notable that the findings of this methodologically robust study on wildfire effects on dissolved organic carbon are different to the majority of other wildfire studies, and there is a need for further studies that employ (B-A-C-I) design to reduce uncertainties surrounding water quality outcomes as a function of wildfire.

There are a number of studies at the UK-scale that assess water quality impacts as a function of managed burning to maintain heather stands, and these have been reviewed by Williamson et al., (2022) with specific reference to dissolved organic matter concentration and quality. Whilst there is evidence to suggest that the effects of managed burning are comparable to wildfires, albeit subdued in both magnitude and duration (Evans et al., 2017; Pilkington et al., 2007; Ramchunder et al., 2013), there should be caution in inferring water quality outcomes between these fire types. Wildfires are more severe, persist for longer and burn deeper than managed fires, and as a result there are likely differences particularly in the post fire erosional potential that could propagate to different water quality responses. Further work is required to understand the comparability of these fire types.

The implications of wildfire for the supply of potable water is of increasing concern (Bladon et al., 2014; Robinne et al., 2021) and emerging water quality issues caused by wildfire have been reported by UK drinking water providers. Most notably, there is a suspected link between wildfire and taste and odour issues reported in drinking water sources, and water companies have observed an increased in customer complaints following wildfires. Ash is hypothesised as a likely contributor to this effect, either as a direct influence or through ash-mobilised contaminant transport to water bodies. However there are a dearth of data on this process and a clear research gap that needs addressing particularly given the current need for drinking water quality providers to understand the impact of climate change on raw water quality.

Longevity of wildfire effects are another source of uncertainty. Short term responses, particularly as a function of high flow events post-fire, have been documented in the literature (e.g. Kettridge et al., 2019), but longer-term effects are less certain. Evidence for effects persisting a few years post-fire is available (e. g. Evans et al., 2017), yet the majority of studies do not monitor much beyond 2 years after the event, and so longer-term effects remain poorly constrained. In a Swedish study stream concentrations of analytes including total organic nitrogen suggested the presence of fast- and slow-release nutrient pools that are attributed to fire physiochemical and biological processes, respectively (Granath et al., 2021), and respond with half-lives of two weeks and four months post-wildfire. The same study found that after three years dissolved nutrient fluxes had returned to pre-fire levels.

3.2. Impacts of wildfire on water quality

There are a number of literature syntheses on the impacts of wildfire on water quality, with provision of potable water and water quality management often the key focus for evidence collation (Bladon et al., 2014; Paul et al., 2022; Smith et al., 2011). However, evidence on the UK scale is generally lacking and therefore knowledge of fire impacts on water quality often must be drawn from international studies, with the majority of evidence gathered in North America (Gomez Isaza et al., 2022). In general, wildfires are considered to release sediment, nutrients and contaminants into the aquatic environment, and change the acidity of the catchment soils resulting in pH shifts in receiving waters. Combined, these water quality changes can have acute and sustained impacts on aquatic ecosystems.

3.2.1. Sediment

Wildfires burn and remove above-ground vegetation, and depending on fire severity, can also burn through soils and below ground organic matter, thereby increasing the risk of soil erosion by several orders of magnitude (Bladon et al., 2014; Smith et al., 2011). Precipitation events following wildfires entrain and suspend eroded material and deliver them to aquatic environments, where increased levels of suspended solids in the water (turbidity) can reduce water clarity and clog stream- or lake-beds with fine particles (Minshall et al., 2001) leading to increased sediment accumulation rates (Evans et al., 2017). Excess loads of sediment to aquatic environments following wildfires can reduce dissolved oxygen resulting in impacts on aquatic biota including fish (Lyon & O'Connor, 2008). The magnitude of the sediment impact is dependent on the severity of the fire, itself determining subsequent vegetation changes and erosion potential.

3.2.2. Nutrients

Burned landscapes have been shown to deliver increased levels of nitrogen, phosphorus, sulfate, chloride, calcium, magnesium, sodium, and potassium to downstream aquatic systems (Bladon et al., 2014). Increases in nitrogen and phosphorus post-wildfire have been observed across ecosystem types and climate zones, and are attributed to reduced plant and microbial uptake and increased leaching into waterbodies (Evans et al., 2017; Granath et al., 2021 Paul et al., 2022; Son et al., 2015). Nitrogen cycling is known to respond to major ecosystem disturbances including wildfires, and for water quality the key effect is sustained loss of NO3- into surrounding drainage systems for a number of years post-fire (Evans et al., 2017; Granath et al., 2021). Fire-induced release of phosphorus can change the dominant form of P speciation in receiving waters (Son et al., 2015) and response magnitudes have generally been shown to be higher for phosphorus than nitrogen (Paul et al., 2022). Combined, excesses of nutrient inputs to aquatic environments can drive enhanced algal growth resulting in negative water quality impacts.

3.2.3. Acidity

Wildfires can change the pH of soil and receiving waters, though there is contrasting evidence regarding whether wildfires increase or decrease pH, and the buffering capacity of both soil and water likely contributes to differences observed between studies. In catchments with low buffering capacity, increases in acid anion concentrations can result in a strong acidification response as observed in a lake catchment post-wildfire by Evans et al., (2017). Alternatively, alkaline responses have been observed due to the increasing the concentration of base cations such as calcium, magnesium, and potassium. For example following a wildfire in Portugal an increase in riverine pH was detected Costa et al., (2014), and this effect persisted for up to a year after the burn. Understanding conditions that may cause contrasting pH responses following a wildfire is important as water pH directly affects aquatic biota and has indirect impacts on pollutant mobilisation (Paul et al., 2022).

3.2.4. Aquatic biota

Few studies have been carried out on the effect of wildfire and subsequent run off into watercourses in the UK. Studies in Portugal demonstrated that chemicals wildfire ash has toxic effects on fish (Nunes et al., 2017). Other studies in USA, Canada and Australia have reported damaging effects of wildfires on aquatic communities, including periphyton and phytoplankton (Charette and Prepas, 2003; Cowell et al., 2006; macroinvertebrates (Earl and Blinn, 2003; Hall and Lombardozzi, 2008; amphibians (Pilliod et al., 2003), and fish (Spencer et al., 2003).

3.3. Key evidence at the UK scale: Wildfire effects on water quality

Given the increasing occurrence of wildfire events in the UK, particularly over the years since 2019, there is a need to understand the likely water quality outcomes of these extreme climatic events. Evidence at the UK scale on wildfire water quality effects is limited and available studies have a bias towards upland peat catchments. A summary of two studies are given below, but there is a need for additional information to determine the water quality risk of wildfires at the UK, with a need for long-term studies across ecosystem types. Recent UK wildfires where the impact on water quality was assessed include the following; 3.3.1 Blue Lough wildfire (2011), 3.3.2 Saddleworth Moor wildfire, 2018, and 3.3.3 the Flow Country wildfire, 2019 and summaries follow.

3.3.1. Case study - Blue Lough wildfire (2011) impact on water quality.

Blue Lough is a small mountain lake that falls within a large (36 km2) drinking water catchment, operated by Northern Ireland Water. On May 1st 2011 the entire Blue Lough catchment was affected by a severe wildfire combusting most above-ground biomass and burning into the organic soil in some areas (Evans et al., 2017). Data from the site collected through the UK Upland Waters Monitoring Network were analysed to determine the wildfire-induced water quality changes. Following the wildfire a large peak in nitrate was detected and it was suggested that a history of high N deposition at Blue Lough had predisposed it to disturbance-induced N loss. Loss of nitrate was detected concurrently with acidification of the lake, detected through pH changes and this finding suggests that UK uplands are sensitive to re-acidifcation through extreme climate events such as wildifres. The Blue Lough wildfire study also produced clear evidence of increased particulate carbon loss, as well as changes in the quality of DOC indicative of a shift towards loss of older soil carbon. Counter to previous studies that suggest that burning leads to a dissolved organic carbon increase, the results from this study show a clear decrease, likely related to changes in organic carbon solubility as a function of wildfire-induced acidification.

3.3.2. Case study - Saddleworth Moor wildfire (2018) impact on water quality.

The 2018 Saddleworth Moor wildfire was one of England's largest recorded, burning over 18 km2 of upland terrain that supplies the greater Manchester area with drinking water. Of particular concern following this fire was the potential release of contaminants into drinking water supplies due to the elevated heavy metal concentrations from past industrial activity Robinne et al., 2021). Whilst there is no peer-reviewed evidence published about this Saddleworth Moor wildfire, a conference presentation by Kettridge et al., (2019) outlined the water quality changes following the first high flow event. There was a dramatic change in pH from 6.4 to 3.6, and there was evidence of ash-bound contaminant mobilisation. Concentrations of Pb peaked during the storm event, but were within previously observed limits of unburned moorland catchments. The water supplier, United Utilities, acted promptly in collaboration with scientists, treating burned hillslopes and gullies with biodegradable erosion prevention measures. This also prompted a study for this region to model contamination potential from future fires in unburned catchments, for future risk mitigation (Robinne et al., 2021).

3.3.3. Case study - Flow Country wildfire (2019) impact on water quality.

Covering 4000km2, the Flow Country is a site of global significance currently under consideration for UNESCO World Heritage Site status. Nevertheless, it has also been substantially modified in places by drainage and notably forestry (with non-native conifer trees), making those areas particularly vulnerable to catastrophic deep burning. These areas have been actively used for scientific research, with a wide range of prior data.

Following a dry and warm spring a large wildfire burnt approximately >60 km2 within the Flow Country peatlands of Caithness and Sutherland, North Scotland in May 2019. Unlike other wildfires in the UK, the May 2019 Flow Country fire covered an exceptionally large area that included peatlands in a range of conditions: drained, drained and afforested, under restoration (through forestry removal and drain blocking) and near-natural. The fire created an urgent and unprecedented opportunity to quantify the interacting effects of fire, drought and past human interventions on peatland carbon storage and water quality. Although fire volatilizes much organic carbon into the atmosphere (Bodi et al., 2014), the effect of this fire in the Flow Country caused higher dissolved organic carbon concentrations in headwater streams in burned drained sites than burned undrained sites (Fernandez et al., 2022), suggesting that drainage primed peatlands for a poorer water quality outcome following the fire. This may be due to burn depth penetrating further into the soil horizon in drained sites. Dissolved organic carbon aromaticity also increased in the burned sites, a response that has been detected in previous studies with implications for drinking water treatment (Williamson et al., 2022). The findings of this study provide some evidence to suggest that peatland restoration though rewetting may reduce impacts of wildfire on water quality.

4. Detection of drought impact on vegetation from Earth Observation data

Glossary of frequently used acronyms

EO = Earth Observation

ET = Evapotranspiration

NDVI = Normalised Difference Vegetation Index

NIR = Near Infrared

RS = Remote sensing

SWIR = Shortwave Infrared

VI = Vegetation index

Remote sensing suggests a recent weakening of the terrestrial carbon sink, identifying notable clusters of browning in forests globally (Winkler et al., 2021). Greening and browning trends reflect changes in the abundance of green leaves, and therefore the rate and amount of photosynthesis. Strengthening of browning trends in tropical biomes is thought to be due to rainfall anomalies (Winkler et al., 2021). This suggests a risk that with increasing drought episodes, tropical forests could switch from a carbon sink to a carbon source (Mitchard, 2018).

The use of remote sensing (RS) data to monitor drought impacts on vegetation has steadily increased in the past half century (West et al., 2019). This growth has accelerated since the early 2000s, probably linked to MODIS (Moderate Resolution Imaging Spectroradiometer) (on board the NASA TERRA and AQUA satellites) which is providing 250m daily global reflectance data since March 2000. A recent review has counted a total of over 60 different EO-based indicators to monitor drought impacts on vegetation (Alahacoon et al., 2022)

4.1. Principles for detecting vegetation drought

The underlying principles for detecting drought impacts on vegetation from EO data are linked to various processes:

Water has an absorption peak in the shortwave infrared (SWIR) spectrum and a reduction in water content in leaves can be directly detected by satellite sensors (Figure below);

Drought causes anatomical changes in leaves, leaf pigment alterations (Figure below) and a reduction in the Leaf Area Index (LAI). This changes the leaf reflectance in the visible and Near infrared spectrum (NIR) and the vegetation indices (VIs) derived from these reflectance signals ;

Once stressed, plants will slow their growth. With severe and prolonged drought, leaves will dry out and fall, ultimately leading to plant mortality. Both processes can be detected using VI time-series;

The lack of water availability and stoma closure will cause a reduction in evapotranspiration (ET), producing a rise in canopy temperature which is visible in the thermal infrared part of the spectrum and the derived land surface temperature (LST) observations;

Reduced photosynthesis (as a consequence of reduced ET and damage to chlorophyll biomolecules) will also lead to a reduction in absorbed photosynthetically active radiation (APAR) and Solar-induced chlorophyll fluorescence (SIF). APAR is derived from the reflectance observed in 400nm-700nm range (Fig.1); SIF is derived from emitted radiation in two very narrow NIR bands 757 nm (758.3–759.2 nm) and 771 nm (769.6–770.3 nm) (Yao et al., 2022).

ET, and indices derived from ET, are closely linked to plant stress, and can be used to estimate drought impacts on vegetation. Algorithms now exist which estimate ET solely from RS data (e.g. GLEAM, Martens et al., 2022);

Agricultural drought is closely linked to soil moisture, which can be estimated using the backscatter signal collected (both passive and active) from the microwave region of the spectrum. Microwave backscatter is sensitive to soil dielectric properties which is influenced by soil water content;

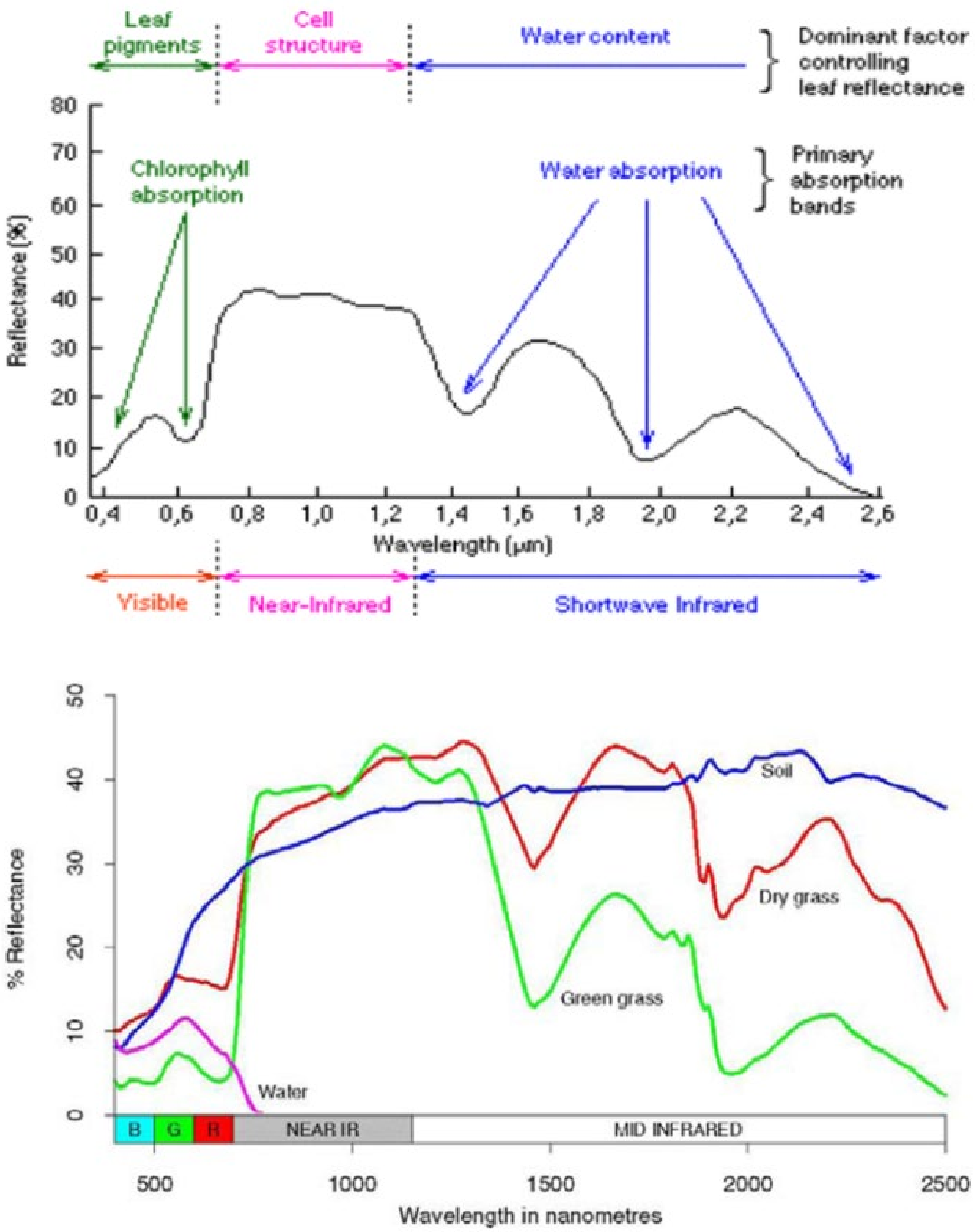


Figure : (a) Spectral Signature of green vegetation, with dominant factor controlling reflectance for different wavelengths, and (b) spectral Signature of green grass (green line) compared to dry grass (red line) and bare soil (blue line). (Image source: from Ashraf et al., 2011).

4.2. Categories of Earth Observation approaches

Based on the principles described above, there broadly are four EO approaches for monitoring drought impacts on vegetation : (i) monitoring changes in biomass and plant water content, using the vegetation's spectral signature in the visible, near- and shortwave infrared spectrum, see Figure above; (ii) monitoring ET, SIF and/or Temperature, using the visible, near- or thermal infrared spectrum; (iii) monitoring soil moisture, using the microwave spectrum, and (iv) increasingly favoured, an integrated approach, combining multiple of these.

4.3. Remote Sensing (RS) products and approaches for vegetation drought impact

West et al., (2019) carried out a systematic review of RS based agricultural drought monitoring since the 70s, and identified the most commonly used RS products:

NDVI (and EVI): The Normalised Difference Vegetation Index (NDVI, Tucker 1979) is the most established approach to vegetation monitoring. Its success derives from the exploitation of the 'red-edge' (sharp increase in vegetation reflectance across the red and NIR regions of the electro-magnetic spectrum, Fig. 1) to detect photosynthetically active plant material, from which plant stress can be inferred. In addition to the NIR and red bands, the blue image band is used to calculate the Enhanced Vegetation Index (EVI, Huete et al., 2002), which is better suited over high biomass regions, as NDVI tends to saturate after a certain point.

GPP: Gross Primary Productivity (GPP) is commonly used to assess vegetation growth and productivity, and represents the rate at which vegetation converts light into energy via photosynthesis Gilabert et al., 2015). It is derived from NDVI, through the light-use efficiency method of Monteith (1972).

Backscatter ratio: The backscatter ratio is the proportion of energy in the microwave region that is reflected back, and is linked to soil moisture and vegetation stress (Vreugdenhil et al., 2018).

VCI: the Vegetation Condition Index (VCI) is a pixel-based normalisation of NDVI, which minimises any spurious or short-term signals in the data, and amplifies the long term trend (Anyamba and Tucker 2012). VCI offers a more robust indicator for seasonal droughts than NDVI (Liu and Kogan 1996). It has been widely used and proved to be effective for monitoring vegetation change and signalling agricultural drought (e.g.Jiao et al., 2016).

NDWI: The Normalised Difference Water Index (NDWI) is very similar to NDVI but uses the NIR and SWIR (instead of red) bands. SWIR is sensitive to soil and vegetation moisture (Gao 1996).

Methods looking at changes in biomass or reduced crop yields using time-series of VIs provide information on the impact of drought and are useful to estimate the damage caused by a drought a posteriori. However, for early warning purposes, and to monitor the development of an agricultural drought, methods that estimate plants' water stress (e.g. based on soil moisture, PET, temperature, leaf water content) are more informative.

More recently, the tendency is to use integrated approaches (Zhang et al., 2017) to build a body of evidence (that includes RS products) showing that the observed vegetation stress is due to a drought event (e.g. change in biomass, ET/temperature, water availability) rather than for example flooding, pests or diseases.

Numerous integrated indices exist (West et al., 2019). One of the most long-standing ones is the Vegetation Health Index (VHI, Kogan 1997), which combines the Vegetation Condition Index (VCI) and the Temperature Condition Index (TCI), and has been used successfully across the world to monitor vegetation stress and drought condition (e.g. Jain et al., 2009, Singh et al., 2003, Unganai and Kogan 1998). Another is the Vegetation Drought Response Index (VegDRI, Brown et al., 2008), which integrates satellite-based observations of vegetation conditions, climate data, and other biophysical information such as land cover/ use, soil characteristics, and ecological setting. It is produced operationally in the U.S. by the National Drought Mitigation Center. Despite showing clear benefits, VegDRI is more complex to compute, with the requirement of many input data, which might explain why its international uptake has been less compared to simpler indicators (Bachmair et al., 2016). Combining VIs with ET data (such as GLEAM) (e.g. Nicolai-Shaw et al., 2017, Orth and Destouni 2018) has also increased in recent years.

The use of machine learning algorithm (AI) and Bayesian inference techniques are currently two rapidly developing areas of research. AI is very effective in finding patterns and connexions within large volumes of multi-source spatio-temporal information, while Bayesian models are well suited for modelling complex spatio-temporal variations and capturing uncertainties. Both have been used to combine RS products with other hydro-climatic data to estimate, provide early warnings or forecast vegetation drought impacts (e.g. for AI: US: Brust et al., (2021), China: Shen et al., (2019), Morocco: Bouras et al., (2021); Australia: Feng et al., (2020); for Bayesian inference: Salakpi et al., (2022). Increasingly also, teams are applying AI methods on data from unmanned aerial vehicles (UAV), or drones, to predict crop yield and monitor drought at field scale (e.g. Kyratis et al., (2010), Ma et al., (2022) Moyer et al., (2023).

4.4. Validation of Earth Observation (EO) products

Many studies internationally have looked at validating these satellite-derived RS drought indicators (especially those based on VIs) with ground-truth data, generally crop yield data, which are used to assess the overall agricultural impact of the drought. For early warning and monitoring on-going droughts however, other field data are more relevant:

some of the commonly used ones for measuring vegetation water content are canopy water content, leaf equivalent water thickness and live fuel moisture content (e.g. Zang et al., 2019). For soil moisture, measurements of volumetric water content with COSMOS instrument (based on Cosmic ray neutron count) have been used successfully to validate RS products (e.g. Peng et al., 2021, Upadhyaya et al., 2021).

Strong correlations were found between drought indicators based on VIs and crop yield for a wide variety of crop types in North America (e.g. U.S., winter wheat, sorghum and corn:Kogan et al., 2012); maize: (Bolton and Freidl 2013), South America (e.g. Brazil, white oat: Coelho et al., 2020), Europe (e.g. Germany, maize: Bachmair et al., 2018; Spain, cereals: Garcia-Leon et al., 2019), Asia (e.g. India, sugarcane: Dubey et al., 2018), the Middle East (e.g. Iran, paddy rice: Esfandabadi et al., 2022), Africa (e.g. Sahelian region, millet and sorghum: Maselli et al., 1995) and Australia (e.g. wheat: Smith et al., 1995). To our knowledge, very few monitoring methods have been applied and tested in temperate, wetter and colder parts of the world such as the UK. One notable exception is the study by Hunt et al., (2019) who showed that combining satellite data (Sentinel-2 MSI) with environmental data and machine learning can accurately predict wheat yield in the UK. However, the focus of the study was not specifically on drought impacts.

4.5. Remote sensing and drought impacts on forests

Remote Sensing to monitor drought impacts on vegetation was originally applied to agriculture, dating back to the 70s (e.g. U.S., wheat: Millard et al., 1978); Israel, wheat: Blum et al., 1982). However, as awareness increased about the important role that forest ecosystems play in global climate, more and more studies have focused on the effect of drought on forests (Zhang et al., 2013, in the boreal (e.g. Reng et al., 2011), Mediterranean (e.g Gouveia et al., 2017) and tropical regions (e.g.Huete et al., 2008), with an explosion of studies in recent years (as shown in review from Torres et al., 2021).

The same or similar indicators used to monitor drought effects on crops are applied on forests (e.g. Buras et al., 2018, Byer and Jin 2017, Marusig et al., 2020 and Miranda et al., 2020). An important difference is that in forests, drought impacts generally appear with a time lag of 1- to 2-month, due to the high soil water holding capacity of forests (Xiao et al., 2005). The effects of drought can also be much more long-lasting than for crops, causing loss of forest biomass and tree mortality, and consequently permanently modifying the structure and dynamics of a forest, leading to changes in the carbon cycle (Allen and Breshears 2010). In some cases, a drought-induced tree mortality has the potential to transform forests from a net carbon sink into a net carbon source (Allen et al., 2010, Yang et al., 2018) either temporarily or permanently. RS can also be used to monitor indirect impacts of droughts on forests, such as insect eruptions (Goodwin et al., 2008, Wulder et al., 2006) and wildfires Justice et al., 2006.

4.6. Remote sensing summary, knowledge gaps and research needs

A huge range of RS-based drought indicators have been developed over the years to monitor drought impacts on vegetation, both for agricultural and forested land. Validation work has demonstrated that these products have great potential to accurately represent drought impacts on the ground. There is a trade-off between simple approaches looking at individual drought effects on plants (loss of biomass, changes in ET or soil moisture), or more complex but more accurate integrated approaches, combining these effects, used in combination with machine learning techniques to predict impacts. However, with the recent proliferation in new approaches it is not obvious which indicator or approach is the most appropriate for operational monitoring purposes. Indicator performance may vary spatially, depending on factors such as crop/forest type, climate, soil type, etc. (Bachmair et al., 2018), making it necessary to carry out local validations. In England and the UK, few studies have assessed indicator performance for operational use. Also, validation data is sparse and only available at field scale.

The main limitation to adequately monitor drought impacts through RS is the necessary trade-off between spatial, temporal and spectral resolution, when all three components are required at high-resolution (West et al., 2019). Sensors with long term records such as the Moderate Resolution Imaging Spectro-radiometer (MODIS, launched in 2000) or National Oceanic and Atmospheric Administration Advanced Very High-Resolution Radiometer (NOAA-AVHRR, first launched in 1978) provide daily data at 1km resolution, and are suitable for regional applications, but too coarse for monitoring at field scale, especially in segmented landscapes such as the British countryside. Other freely available sensors such as MSS, TM, ETM and OLI on board the Landsat series (first launched in 1972) offer data at high-resolution (30m) every 16 days which generally results in a limited number of useable (cloud free) observations per year. This is not enough for monitoring and early warning purposes (Graw et al., 2019). The recent Sentinel-2 MSI mission (first launched in 2015) with a revisit time of 5 days and spatial resolution of up to 10m is enabling improvements in drought impact monitoring and initial studies have suggested a real potential in the remote sensing of vegetation (e.g. Lambert et al., 2018, Sadeghi et al., 2017, Vanino et al., 2018).

Progressing the use of RS for drought impact monitoring and forecasting in England and the UK will require solutions that reduce the impact of cloud on observation availability and explore the use of AI and/or Bayesian inference approaches, and an increase in the spatial and temporal volume of validation data.

5. Using Earth System Models to understand drought impacts

Drought often co-occurs with other stresses such as high temperatures, elevated ozone concentrations, rising CO2 concentrations and increased Nitrogen deposition. This complexity of interacting effects varies over time and between geographic location, due to, for example, heterogeneity of soil properties, elevation, and vegetation cover. The net result of these interactions at the global scale determines the amount of carbon taken up and stored by the land. Understanding the size, spatial heterogeneity, and persistence of the carbon sink now and in the future is important because of its role in driving climate-carbon feedbacks (where uptake of CO2 by plants slows the growth rate of CO2 in the atmosphere, therefore slowing global warming). It also controls surface energy exchanges that impact regional temperature and precipitation.

Vegetation responses to interactions between different climate and environmental drivers, and how the response of the land surface feeds back on to climate can be explored using Earth System Models (ESMs). The land surface component of the ESM simulates the exchange of carbon, water and energy between the land surface and the atmosphere. This land surface component (also referred to as a Land Surface Model (LSM) therefore provides the lower boundary conditions for the atmospheric component of the ESM. ESMs/LSMs allow exploration of the vegetation response to changing environmental conditions over large spatial scales covering changing gradients of soil, climate and other environmental conditions, and over long temporal scales to investigate historical and future impacts under different scenarios of climate change. ESM projections form the main tool to predict future climate change and underpin much of the regular United Nations Intergovernmental Panel on Climate Change (IPCC) reports that inform policymakers.

However, high uncertainty in ESM predictions of carbon cycle dynamics is well known (Arora et al., 2013) owing to the complex nature of the biological processes involved. Model predictions of the terrestrial carbon store range from a significant carbon sink to a carbon source (Cox et al., 2013, Sitch et al., 2015). This variability remains an issue, with the large spread seen across CMIP5 land carbon cycle models little changed in CMIP6 (Arora et al., 2020).

5.1. Knowledge gaps For Earth System and Land Surface models

The large uncertainties in Earth System Model predictions of the future global carbon sink are due, in part, to poor/missing representation of key processes such as how vegetation responds to water stress. For example, ESMs fail to capture mortality in trees due to drought, and acclimation of trees to drought including changes in biomass partitioning. Prediction is difficult because the physiological mechanisms underlying drought survival and mortality remain poorly understood, nevertheless, ESMs lack key processes that would improve their ability to predict ecosystem responses to drought (McDowell et al., 2008). Climate model predictions suggest severe and widespread droughts over many regions of the globe in the next 30 - 90 years (Dai, 2013), which suggests improving the ability of EMSs to model vegetation response to water stress should be a priority.

In most Land Surface Models the treatment of soil moisture stress on vegetation is through a linear response function (β) that limits gas exchange as water deficit increases (Verhoef & Egea 2014). The use of the β function has been identified as a key source of uncertainty in LSMs (Blyth et al., 2011, Verhoef and Egea, 2014, Vidale et al., 2021). Currently, ESMs tend to represent plant water stress by either reducing stomatal conductance or photosynthetic capacity or both (De Kauwe et al., 2015). Vidale et al., (2021) explored combinations of non-linear β function responses applied at different points in the photosynthesis – stomatal conductance pathway (i.e. carbon assimilation, stomatal conductance, or mesophyll conductance). They found that treatments allowing ß to act on vegetation fluxes via stomatal and mesophyll routes were better able to capture the spatiotemporal variability in water use efficiency during the growing season, and therefore improved responses to water stressed conditions. However, whilst alternative parameterisations of β are a first step to improving modelled plant responses to water deficit, further development to how the soil-plant hydraulic system is represented is required to capture plant water dynamics more accurately. The use of optimality-based plant hydraulic transport models shows some promise (Eller et al., 2020).

The response of photosynthesis to temperature has been identified as a significant source of uncertainty in Ecosystem models (Rogers et al., 2017, Tharammal et al., 2019). Most models represent the optimal temperature for photosynthesis as fixed in both space and time, differentiated only by photosynthetic pathway (C3/C4), or by coarse plant functional type categories.

References collated by section (some references included for completeness here and as a future resource are not included in the text)

References: Section 1 Introduction and scope, plants, vegetation and drought. Including section 5 Using Earth System Models to understand drought impacts:

Alves, R. D. F. B., Menezes-Silva, P. E., Sousa, I. F., Loram-Lourenço, I., Silva, M. L. F., Almeida, S. E. S., Silva, F. G., Perez de Souza, I., Fernie, A. R. & Farnese, F. S. 2020. Evidence of drought memory in Dipteryx alata indicates differential acclimation of plants to savanna conditions. Scientific Reports, 10, 16455.

Ammer, C. 2019. Diversity and forest productivity in a changing climate. New Phytologist, 221, 50-66.

Anderegg, W. R. L., Konings, A. G., Trugman, A. T., Yu, K., Bowling, D. R., Gabbitas, R., Karp, D. S., Pacala, S., Sperry, J. S., Sulman, B. N. & Zenes, N. 2018. Hydraulic diversity of forests regulates ecosystem resilience during drought. Nature, 561, 538-541.

Anderegg, William R. L. Joseph A. Berry, Duncan D. Smith, John S. Sperry, Leander D. L. Anderegg, and Christopher B. Field (2011) The roles of hydraulic and carbon stress in a widespread climate-induced forest die-off. PNAS 109 (1) 233-237 https://doi.org/10.1073/pnas.1107891109

ARORA, V. K., BOER, G. J., FRIEDLINGSTEIN, P., EBY, M., JONES, C. D., CHRISTIAN, J. R., BONAN, G., BOPP, L., BROVKIN, V., CADULE, P., HAJIMA, T., ILYINA, T., LINDSAY, K., TJIPUTRA, J. F. & WU, T. 2013. Carbon–Concentration and Carbon–Climate Feedbacks in CMIP5 Earth System Models. Journal of Climate, 26, 5289-5314.

ARORA, V. K., KATAVOUTA, A., WILLIAMS, R. G., JONES, C. D., BROVKIN, V., FRIEDLINGSTEIN, P., SCHWINGER, J., BOPP, L., BOUCHER, O., CADULE, P., CHAMBERLAIN, M. A., CHRISTIAN, J. R., DELIRE, C., FISHER, R. A., HAJIMA, T., ILYINA, T., JOETZJER, E., KAWAMIYA, M., KOVEN, C. D., KRASTING, J. P., LAW, R. M., LAWRENCE, D. M., LENTON, A., LINDSAY, K., PONGRATZ, J., RADDATZ, T., SÉFÉRIAN, R., TACHIIRI, K., TJIPUTRA, J. F., WILTSHIRE, A., WU, T. & ZIEHN, T. 2020. Carbon–concentration and carbon–climate feedbacks in CMIP6 models and their comparison to CMIP5 models. Biogeosciences, 17, 4173-4222.

Ayling S.M. J. Thompson, A. Gray. L.J. McEwan 2021 Impact of Reduced Rainfall on Above Ground Dry Matter Production of Semi-natural Grassland in South Gloucestershire, UK: A Rainfall Manipulation Study Frontiers in Env Sci Volume 9 https://doi.org/10.3389/fenvs.2021.686668

Barros, F. d. V., Bittencourt, P. R. L., Brum, M., Restrepo-Coupe, N., Pereira, L., Teodoro, G. S., Saleska, S. R., Borma, L. S., Christoffersen, B. O., Penha, D., Alves, L. F., Lima, A. J. N., Carneiro, V. M. C., Gentine, P., Lee, J.-E., Aragão, L. E. O. C., Ivanov, V., Leal, L. S. M., Araujo, A. C., & Oliveira, R. S. (2019). Hydraulic traits explain differential responses of Amazonian forests to the 2015 El Niño-induced drought. The New Phytologist, 223, 1253–1266. In Zuleta 2022

Bashir, Sheikh Shanawaz, Anjuman Hussain, Sofi Javed Hussain, Owais Ali Wani, Sheikh Zahid Nabi, Niyaz A. Dar, Faheem Shehzad Baloch & Sheikh Mansoor (2021) Plant drought stress tolerance: understanding its physiological, biochemical and molecular mechanisms, Biotechnology & Biotechnological Equipment, 35:1, 1912-1925,DOI:10.1080/13102818.2021.2020161

BATTIPAGLIA, J. G., BRIENEN, R. J. W., CABUGAO, K. G., CAILLERET, M., CAMPBELL, E., CANADELL, J. G., CIAIS, P., CRAIG, M. E., ELLSWORTH, D. S., FARQUHAR, G. D., FATICHI, S., FISHER, J. B., FRANK, D. C., GRAVEN, H., GU, L., HAVERD, V., HEILMAN, K., HEIMANN, M., HUNGATE, B. A., IVERSEN, C. M., JOOS, F., JIANG, M., KEENAN, T. F., KNAUER, J., KÖRNER, C., LESHYK, V. O., LEUZINGER, S., LIU, Y., MACBEAN, N., MALHI, Y., MCVICAR, T. R., PENUELAS, J., PONGRATZ, J., POWELL, A. S., RIUTTA, T., SABOT, M. E. B., SCHLEUCHER, J., SITCH, S., SMITH, W. K., SULMAN, B., TAYLOR, B., TERRER, C., TORN, M. S., TRESEDER, K. K., TRUGMAN, A. T., TRUMBORE, S. E., VAN MANTGEM, P. J., VOELKER, S. L., WHELAN, M. E. & ZUIDEMA, P. A. 2021. Integrating the evidence for a terrestrial carbon sink caused by increasing atmospheric CO2. New Phytologist, 229, 2413-2445.

Basu S, Ramegowda V, Kumar A and Pereira A. Plant adaptation to drought stress [version 1; peer review: 3 approved]. F1000Research 2016, 5(F1000 Faculty Rev):1554 (https://doi.org/10.12688/f1000research.7678.1)

Benjamin Lloyd-Hughes (2014) The impracticality of a universal drought definition. Theor Appl Climatol (2014) 117:607–611 DOI 10.1007/s00704-013-1025-7

Bowles TM, Atallah SS, Campbell EE, Gaudin AC, Wieder WR, Grandy AS (2018) Addressing agricultural nitrogen losses in a changing climate. Nat Sustain 1:399

Bohnert HJ, Nelson DE, Jensen RG: Adaptations to Environmental Stresses. Plant Cell. 1995; 7(7): 1099–111.   In The physiology of plant responses to drought Aditi Gupta ,AndrésRico-Medina and Ana i. Caño-Delgado (2020) Science Vol 368, Issue 6488 pp. 266-269 DOI: 10.1126/science.aaz7614

Brundrett MC, Tedersoo L (2018) Evolutionary history of mycorrhizal symbioses and global host plant diversity. New Phytol 220:1108–1115

Bradstock, R. A. A biogeographic model of fire regimes in Australia: current and future implications. Glob. Ecol. Biogeogr. 19, 145–158 (2010).

BLYTH, E., CLARK, D. B., ELLIS, R., HUNTINGFORD, C., LOS, S., PRYOR, M., BEST, M. & SITCH, S. 2011. A comprehensive set of benchmark tests for a land surface model of simultaneous fluxes of water and carbon at both the global and seasonal scale. Geosci. Model Dev., 4, 255-269.

BRIENEN, R. J. W., PHILLIPS, O. L., FELDPAUSCH, T. R., GLOOR, E., BAKER, T. R., LLOYD, J., LOPEZ-GONZALEZ, G., MONTEAGUDO-MENDOZA, A., MALHI, Y., LEWIS, S. L., VÁSQUEZ MARTINEZ, R., ALEXIADES, M., ÁLVAREZ DÁVILA, E., ALVAREZ-LOAYZA, P., ANDRADE, A., ARAGÃO, L. E. O. C., ARAUJO-MURAKAMI, A., ARETS, E. J. M. M., ARROYO, L., AYMARD C, G. A., BÁNKI, O. S., BARALOTO, C., BARROSO, J., BONAL, D., BOOT, R. G. A., CAMARGO, J. L. C., CASTILHO, C. V., CHAMA, V., CHAO, K. J., CHAVE, J., COMISKEY, J. A., CORNEJO VALVERDE, F., DA COSTA, L., DE OLIVEIRA, E. A., DI FIORE, A., ERWIN, T. L., FAUSET, S., FORSTHOFER, M., GALBRAITH, D. R., GRAHAME, E. S., GROOT, N., HÉRAULT, B., HIGUCHI, N., HONORIO CORONADO, E. N., KEELING, H., KILLEEN, T. J., LAURANCE, W. F., LAURANCE, S., LICONA, J., MAGNUSSEN, W. E., MARIMON, B. S., MARIMON-JUNIOR, B. H., MENDOZA, C., NEILL, D. A., NOGUEIRA, E. M., NÚÑEZ, P., PALLQUI CAMACHO, N. C., PARADA, A., PARDO-MOLINA, G., PEACOCK, J., PEÑA-CLAROS, M., PICKAVANCE, G. C., PITMAN, N. C. A., POORTER, L., PRIETO, A., QUESADA, C. A., RAMÍREZ, F., RAMÍREZ-ANGULO, H., RESTREPO, Z., ROOPSIND, A., RUDAS, A., SALOMÃO, R. P., SCHWARZ, M., SILVA, N., SILVA-ESPEJO, J. E., SILVEIRA, M., STROPP, J., TALBOT, J., TER STEEGE, H., TERAN-AGUILAR, J., TERBORGH, J., THOMAS-CAESAR, R., TOLEDO, M., TORELLO-RAVENTOS, M., UMETSU, R. K., VAN DER HEIJDEN, G. M. F., VAN DER HOUT, P., GUIMARÃES VIEIRA, I. C., VIEIRA, S. A., VILANOVA, E., VOS, V. A. & ZAGT, R. J. 2015.. Long-term decline of the Amazon carbon sink. Nature, 519, 344-348.

BROECKX, L. S., VERLINDEN, M. S., BERHONGARAY, G., ZONA, D., FICHOT, R. & CEULEMANS, R. 2014. The effect of a dry spring on seasonal carbon allocation and vegetation dynamics in a poplar bioenergy plantation. GCB Bioenergy, 6, 473-487.

Buchenau, N., van Kleunen, M. and Wilschut, R.A. (2022), Direct and legacy-mediated drought effects on plant performance are species-specific and depend on soil community composition. Oikos, 2022: e08959. https://doi.org/10.1111/oik.08959

Carla Vázquez-González; Laura Pombo-Salinas; Lucía Martín-Cacheda; Sergio Rasmann; Gregory Röder; Luis Abdala-Roberts; Kailen A. Mooney; Xoaquín Moreira 2022 Functional Ecology 36:11 2763-2773 DOI: 10.1111/1365-2435.14159

Gaspar MJ, Louzada JL, Silva ME, Aguiar A, Almeida MH (2008) Age trends in genetic parameters of wood density components in 46 half-sibling families of Pinus pinaster. Can J For Res 38:1470–1477

CHAVES, M. M., PEREIRA, J. S., MAROCO, J., RODRIGUES, M. L., RICARDO, C. P. P., OSORIO, M. L., CARVALHO, I., FARIA, T. & PINHEIRO, C. 2002. How plants cope with water stress in the field? Photosynthesis and growth. Annals Botany, 89, 907-916.

CHOAT, B., BRODRIBB, T. J., BRODERSEN, C. R., DUURSMA, R. A., LÓPEZ, R. & MEDLYN, B. E. (2018). Triggers of tree mortality under drought. Nature, 558, 531-539. https://doi.org/10.1038/s41586-018-0240-x

CHOAT, B., JANSEN, S., BRODRIBB, T. J., COCHARD, H., DELZON, S., BHASKAR, R., BUCCI, S. J., FEILD, T. S., GLEASON, S. M., HACKE, U. G., JACOBSEN, A. L., LENS, F., MAHERALI, H., MARTÍNEZ-VILALTA, J., MAYR, S., MENCUCCINI, M., MITCHELL, P. J., NARDINI, A., PITTERMANN, J., PRATT, R. B., SPERRY, J. S., WESTOBY, M., WRIGHT, I. J. & ZANNE, A. E. 2012. Global convergence in the vulnerability of forests to drought. Nature, 491, 752-755.

CLIFTON-BROWN, J. C. & LEWANDOWSKI, I. 2000. Water use efficiency and biomass partitioning of three different Miscanthus genotypes with limited and unlimited water supply. Annals Botany, 86, 191-200.

CLIFTON-BROWN, J. C., LEWANDOWSKI, I. & JONES, M. B. 2002. Comparative responses to water stress in stay-green, rapid- and slow senescing genotypes of the biomass crop, Miscanthus. New Phytologist, 154, 335-345.

Correia, A.H., Almeida, M.H., Branco, M., Tomé, M., Cordero Montoya, R., Di Lucchio, L., Cantero, A., Diez, J.J., Prieto-Recio, C., Bravo, F. and Gartzia, N., 2018. Early survival and growth plasticity of 33 species planted in 38 arboreta across the European Atlantic area. Forests, 9(10), p.630.

COX, P. M., PEARSON, D., BOOTH, B. B., FRIEDLINGSTEIN, P., HUNTINGFORD, C., JONES, C. D. & LUKE, C. M. 2013. Sensitivity of tropical carbon to climate change constrained by carbon dioxide variability. Nature, 494, 341-344.

Csilléry, K., Ovaskainen, O., Sperisen, C. et al., Adaptation to local climate in multi-trait space: evidence from silver fir (Abies alba Mill.) populations across a heterogeneous environment. Heredity 124, 77–92 (2020). https://doi.org/10.1038/s41437-019-0240-0

DAI, A. 2013. Increasing drought under global warming in observations and models. Nature Climate Change, 3, 52-58.

Daryanto S., Wang L., Jacinthe P.A. Global synthesis of drought effects on cereal, legume, tuber and root crops production: A review. Agric. Water Manag. 2020;179:18–33. doi: 10.1016/j.agwat.2016.04.022.

DE ARELLANO, J. V.-G., VAN HEERWAARDEN, C. C. & LELIEVELD, J. 2012. Modelled suppression of boundary-layer clouds by plants in a CO2-rich atmosphere. Nature Geoscience, 5, 701-704.

DE KAUWE, M. G., MEDLYN, B. E., ZAEHLE, S., WALKER, A. P., DIETZE, M. C., WANG, Y.-P., LUO, Y., JAIN, A. K., EL-MASRI, B., HICKLER, T., WÅRLIND, D., WENG, E., PARTON, W. J., THORNTON, P. E., WANG, S., PRENTICE, I. C., ASAO, S., SMITH, B., MCCARTHY, H. R., IVERSEN, C. M., HANSON, P. J., WARREN, J. M., OREN, R. & NORBY, R. J. 2014. Where does the carbon go? A model–data intercomparison of vegetation carbon allocation and turnover processes at two temperate forest free-air CO2 enrichment sites. New Phytologist, 203, 883-899.

DE KAUWE, M. G., ZHOU, S. X., MEDLYN, B. E., PITMAN, A. J., WANG, Y. P., DUURSMA, R. A. & PRENTICE, I. C. 2015. Do land surface models need to include differential plant species responses to drought? Examining model predictions across a mesic-xeric gradient in Europe. Biogeosciences, 12, 7503-7518.

DESOTO, L., CAILLERET, M., STERCK, F., JANSEN, S., KRAMER, K., ROBERT, E. M. R., AAKALA, T., AMOROSO, M. M., BIGLER, C., CAMARERO, J. J., ČUFAR, K., GEA-IZQUIERDO, G., GILLNER, S., HAAVIK, L. J., HEREŞ, A.-M., KANE, J. M., KHARUK, V. I., KITZBERGER, T., KLEIN, T., LEVANIČ, T., LINARES, J. C., MÄKINEN, H., OBERHUBER, W., PAPADOPOULOS, A., ROHNER, B., SANGÜESA-BARREDA, G., STOJANOVIC, D. B., SUÁREZ, M. L., VILLALBA, R. & MARTÍNEZ-VILALTA, J. 2020. Low growth resilience to drought is related to future mortality risk in trees. Nature Communications, 11, 545.

Desprez-Loustau, M.L., Marçais, B., Nageleisen, L.M., Piou, D. and Vannini, A., 2006. Interactive effects of drought and pathogens in forest trees. Annals of forest science, 63(6), pp.597-612.

DREWNIAK, B. & GONZALEZ-MELER, M. A. 2017. Earth System Model Needs for Including the Interactive Representation of Nitrogen Deposition and Drought Effects on Forested Ecosystems. Forests, 8, 267.

DUSENGE, M. E., MADHAVJI, S. & WAY, D. A. 2020. Contrasting acclimation responses to elevated CO2 and warming between an evergreen and a deciduous boreal conifer. Global Change Biology, 26, 3639-3657.

ELLER, C. B., ROWLAND, L., MENCUCCINI, M., ROSAS, T., WILLIAMS, K., HARPER, A., MEDLYN, B. E., WAGNER, Y., KLEIN, T., TEODORO, G. S., OLIVEIRA, R. S., MATOS, I. S., ROSADO, B. H. P., FUCHS, K., WOHLFAHRT, G., MONTAGNANI, L.,

MEIR, P., SITCH, S. & COX, P. M. 2020. Stomatal optimization based on xylem hydraulics (SOX) improves land surface model simulation of vegetation responses to climate. New Phytologist, 226, 1622-1637.

Fang Y, Xiong L (2015) General mechanisms of drought response and their application in drought resistance improvement in plants. Cell Mol Life Sci 72:673–689

FLEXAS, J., BOTA, J., GALMES, J., MEDRANO, H. & RIBAS-CARBO, M. 2006. Keeping a positive carbon balance under adverse conditions: responses of photosynthesis and respiration to water stress. Physiologia Plantarum, 127, 343-352.

FRANKLIN, O., JOHANSSON, J., DEWAR, R. C., DIECKMANN, U., MCMURTRIE, R. E., BRÄNNSTRÖM, A. K. & DYBZINSKI, R. 2012. Modeling carbon allocation in trees: A search for principles. Tree Physiol, 32, 648–666.

FRANKS, P. J., BERRY, J. A., LOMBARDOZZI, D. L. & BONAN, G. B. 2017. Stomatal Function across Temporal and Spatial Scales: Deep-Time Trends, Land-Atmosphere Coupling and Global Models. Plant Physiology, 174, 583-602.

FRANKS, P. J., BONAN, G. B., BERRY, J. A., LOMBARDOZZI, D. L., HOLBROOK, N. M., HEROLD, N. & OLESON, K. W. 2018. Comparing optimal and empirical stomatal conductance models for application in Earth system models. Global Change Biology, 24, 5708-5723.

FRIEDLINGSTEIN, P., JONES, M. W., O'SULLIVAN, M., ANDREW, R. M., HAUCK, J., PETERS, G. P., PETERS, W., PONGRATZ, J., SITCH, S., LE QUÉRÉ, C., BAKKER, D. C. E., CANADELL, J. G., CIAIS, P., JACKSON, R. B., ANTHONI, P., BARBERO, L., BASTOS, A., BASTRIKOV, V., BECKER, M., BOPP, L., BUITENHUIS, E., CHANDRA, N., CHEVALLIER, F., CHINI, L. P., CURRIE, K. I., FEELY, R. A., GEHLEN, M., GILFILLAN, D., GKRITZALIS, T., GOLL, D. S., GRUBER, N., GUTEKUNST, S., HARRIS, I., HAVERD, V., HOUGHTON, R. A., HURTT, G., ILYINA, T., JAIN, A. K., JOETZJER, E., KAPLAN, J. O., KATO, E., KLEIN GOLDEWIJK, K., KORSBAKKEN, J. I., LANDSCHÜTZER, P., LAUVSET, S. K., LEFÈVRE, N., LENTON, A., LIENERT, S., LOMBARDOZZI, D., MARLAND, G., MCGUIRE, P. C., MELTON, J. R., METZL, N., MUNRO, D. R., NABEL, J. E. M. S., NAKAOKA, S. I., NEILL, C., OMAR, A. M., ONO, T., PEREGON, A., PIERROT, D., POULTER, B., REHDER, G., RESPLANDY, L., ROBERTSON, E., RÖDENBECK, C., SÉFÉRIAN, R., SCHWINGER, J., SMITH, N., TANS, P. P., TIAN, H., TILBROOK, B., TUBIELLO, F. N., VAN DER WERF, G.

Gontia-Mishra I., Sapre S., Sharma A., Tiwari S. Amelioration of drought tolerance in wheat by the interaction of plant growth-promoting rhizobacteria. Plant Biol.2016;18:992–1000. doi: 10.1111/plb.12505.

Grams, T. E. E., Hesse, B. D., Gebhardt, T., Weikl, F., Rötzer, T., Kovacs, B., Hikino, K., Hafner, B. D., Brunn, M., Bauerle, T., Häberle, K.-H., Pretzsch, H., and Pritsch, K.. 2021.

The Kroof experiment: realization and efficacy of a recurrent drought experiment plus recovery in a beech/spruce forest. Ecosphere 12(3):e03399. 10.1002/ecs2.3399

Grime J, Fridley JD, Askew AP, Thompson K, Hodgson JG, Bennett CR. 2008. Long-term resistance to simulated climate change in an infertile grassland. Proc Natl Acad Sci USA 105:10028–32

HACKE, U. G., SPERRY, J. S., POCKMAN, W. T., DAVIS, S. D. & MCCULLOH, K. A. 2001. Trends in wood density and structure are linked to prevention of xylem implosion by negative pressure. Oecologia, 126, 457-461.

HARRISON, S. P., CRAMER, W., FRANKLIN, O., PRENTICE, I. C., WANG, H., BRÄNNSTRÖM, Å., DE BOER, H., DIECKMANN, U., JOSHI, J. & KEENAN, T. F. 2021. Eco-evolutionary optimality as a means to improve vegetation and land-surface models. New Phytol, 231, 2125-2141.

Haugwitz MS, Bergmark L, Prieme´ A, Christensen S, Beier C, Michelsen A. 2014. Soil microorganisms respond to five years of climate change manipulations and elevated atmospheric CO2 in a temperate heath ecosystem. Plant Soil 374:211–222

HERTEL, D., STRECKER, T., MÜLLER-HAUBOLD, H. & LEUSCHNER, C. 2013. Fine root biomass and dynamics in beech forests across a precipitation gradient – is optimal resource partitioning theory applicable to water-limited mature trees? Journal of Ecology, 101, 1183-1200.

Hossain, M.L., J. Li, S. Hoffmann and C. Beierkuhnlein (2022). Biodiversity showed positive effects on resistance but mixed effects on resilience to climatic extremes in a long-term grassland experiment. Science of The Total Environment 2022 Vol. 827 Pages 154322. DOI: https://doi.org/10.1016/j.scitotenv.2022.154322

Jones, M. W., Smith, A. J. P., Betts, R., Canadell, J. G., Prentice, I. C., and Le Quere,

C. (2020). Climate Change Increases the Risk of Wildfires: January 2020. ScienceBrief. https://news.sciencebrief.org/wildfires-sep2020-update/ accessed 22 March 2023

Ilyas, M., Nisar, M., Khan, N. et al., Drought Tolerance Strategies in Plants: A Mechanistic Approach. J Plant Growth Regul 40, 926–944 (2021). https://doi.org/10.1007/s00344-020-10174-5

Jactel, H., Petit, J., Desprez-Loustau, M.L., Delzon, S., Piou, D., Battisti, A. and Koricheva, J., 2012. Drought effects on damage by forest insects and pathogens: a meta-analysis. Global Change Biology, 18(1), pp.267-276.

Jaideep Joshi1, Benjamin D. Stocker, Florian Hofhansl5, Shuangxi Zhou, Ulf Dieckmann, and Iain Colin Prentice (2023). Towards a unified theory of plant photosynthesis and hydraulics bioRxiv preprint https://doi.org/10.1101/2020.12.17.423132; https://www.biorxiv.org/content/10.1101/2020.12.17.423132v2.full.pdf version 2 accessed 23 Jan 2023

JASECHKO, S., SHARP, Z. D., GIBSON, J. J., BIRKS, S. J., YI, Y. & FAWCETT, P. J. 2013. Terrestrial water fluxes dominated by transpiration. Nature, 496, 347-350.

JIANG, C., RYU, Y., WANG, H. & KEENAN, T. F. 2020. An optimality-based model explains seasonal variation in C3 plant photosynthetic capacity. GCB, 26, 6493-6510.

Jump, A.S., Ruiz-Benito, P., Greenwood, S., Allen, C.D., Kitzberger, T., Fensham, R., Martínez-Vilalta, J. and Lloret, F., 2017. Structural overshoot of tree growth with climate variability and the global spectrum of drought-induced forest dieback. Global change biology, 23(9), pp.3742-3757

KALA, J., DE KAUWE, M. G., PITMAN, A. J., LORENZ, R., MEDLYN, B. E., WANG, Y. P., LIN, Y. S. & ABRAMOWITZ, G. 2015. Implementation of an optimal stomatal conductance scheme in the Australian Community Climate Earth Systems Simulator (ACCESS1.3b). Geosci. Model Dev., 8, 3877-3889.

KALA, J., DE KAUWE, M. G., PITMAN, A. J., MEDLYN, B. E., WANG, Y.-P., LORENZ, R. & PERKINS-KIRKPATRICK, S. E. 2016. Impact of the representation of stomatal conductance on model projections of heatwave intensity. Scientific Reports, 6, 23418.

KOOPERMAN, G. J., CHEN, Y., HOFFMAN, F. M., KOVEN, C. D., LINDSAY, K., PRITCHARD, M. S., SWANN, A. L. S. & RANDERSON, J. T. 2018. Forest response to rising CO2 drives zonally asymmetric rainfall change over tropical land. Nature Climate Change, 8, 434-440.

KATTGE, J. & KNORR, W. 2007. Temperature acclimation in a biochemical model of photosynthesis: a reanalysis of data from 36 species. Plant, Cell & Environment, 30, 1176-1190.

KUMARATHUNGE, D. P., MEDLYN, B. E., DRAKE, J. E., TJOELKER, M. G., ASPINWALL, M. J., BATTAGLIA, M., CANO, F. J., CARTER, K. R., CAVALERI, M. A., CERNUSAK, L. A., CHAMBERS, J. Q., CROUS, K. Y., DE KAUWE, M. G., DILLAWAY, D. N., DREYER, E., ELLSWORTH, D. S., GHANNOUM, O., HAN, Q., HIKOSAKA, K., JENSEN, A. M., KELLY, J. W. G., KRUGER, E. L., MERCADO, L. M., ONODA, Y., REICH, P. B., ROGERS, A., SLOT, M., SMITH, N. G., TARVAINEN, L., TISSUE, D. T., TOGASHI, H. F., TRIBUZY, E. S., UDDLING, J., VÅRHAMMAR, A., WALLIN, G., WARREN, J. M. & WAY, D. A. 2019. Acclimation and adaptation components of the temperature dependence of plant photosynthesis at the global scale. New Phytologist, 222, 768-784.

Kannenberg, S.A., Phillips, R.P. Soil microbial communities buffer physiological responses to drought stress in three hardwood species. Oecologia 183, 631–641 (2017). https://doi.org/10.1007/s00442-016-3783-2

Keith, D. A., Holman, L., Rodoreda, S., Lemmon, J. & Bedward, M. Plant functional types can predict decade-scale changes in fire-prone vegetation. J. Ecol. 95, 1324–1337 (2007).

Khalil, A. A. M., and Grace, J. (1992). Acclimation to Drought in Acer Pseudoplatanus L. (Sycamore) Seedlings. J. Exp. Bot. 43 (257), 1591–1602. doi:10.1093/jxb/43.12.1591

Koen Kramer (1995) Modelling comparison to evaluate the importance of phenology for the effects of climate change on growth of temperate-zone deciduous trees Climate Research, Vol. 5, No. 2 (June 22, 1995), pp. 119-130

Kour, D., Yadav, A.N. Bacterial Mitigation of Drought Stress in Plants: Current Perspectives and Future Challenges. Curr Microbiol 79, 248 (2022). https://doi.org/10.1007/s00284-022-02939-w

Kongstad, J., Schmidt, I.K., Riis-Nielsen, T. et al., High Resilience in Heathland Plants to Changes in Temperature, Drought, and CO2 in Combination: Results from the CLIMAITE Experiment. Ecosystems 15, 269–283 (2012). https://doi.org/10.1007/s10021-011-9508-9

Kramer K (1995) Phenotypic plasticity of the phenology of seven European tree species in relation to climatic warming Plant Cell Environ 18:93-104

Kunstler G, Albert CH, Courbaud B et al., (2011) Effects of competition on tree radial-growth vary in importance but not in intensity along climatic gradients. J Ecol 99:300–312

LAWLOR, D. W. & CORNIC, G. 2002. Photosynthetic carbon assimilation and associated metabolism in relation to water deficits in higher plants. Plant , Cell and Environment, 25, 275-294.

LENS, F., SPERRY, J. S., CHRISTMAN, M. A., CHOAT, B., RABAEY, D. & JANSEN, S. 2011. Testing hypotheses that link wood anatomy to cavitation resistance and hydraulic conductivity in the genus Acer. New Phytologist, 190, 709-723.

LEWIS, S. L., BRANDO, P. M., PHILLIPS, O. L., VAN DER HEIJDEN, G. M. F. & NEPSTAD, D. 2011. The 2010 Amazon Drought. Science, 331, 554-554.

LIMOUSIN, J.-M., MISSON, L., LAVOIR, A.-V., MARTIN, N. K. & RAMBAL, S. 2010. Do photosynthetic limitations of evergreen Quercus ilex leaves change with long-term increased drought severity? Plant, Cell & Environment, 33, 863-875.

Locatelli, Tom, Kate Beauchamp, Mike Perks, Georgios Xenakis, Bruce Nicoll, James Morison, Forest Research February 2021 ClimateXchange publications   DOI: https://doi.org/10.7488/era/1292 accessed 23 March 2023 https://era.ed.ac.uk/handle/1842/36757

LO GULLO, M.A. and SALLEO, S. (1988), Different strategies of drought resistance in three Mediterranean sclerophyllous trees growing in the same environmental conditions. New Phytologist, 108: 267-276. https://doi.org/10.1111/j.1469-8137.1988.tb04162.x

LOMBARDOZZI, D. L., BONAN, G. B., SMITH, N. G., DUKES, J. S. & FISHER, R. A. 2015. Temperature acclimation of photosynthesis and respiration: A key uncertainty in the carbon cycle-climate feedback. Geophysical Research Letters, 42, 8624-8631.

Luo LJ (2010) Breeding for water-saving and drought-resistance rice (WDR) in China. J Exp Bot 61:3509–3517

Madouh Tareq A. (2022) Eco-Physiological Responses of Native Desert Plant Species to Drought and Nutritional Levels: Case of Kuwait Frontiers in Environmental Science 10 2022   URL: https://www.frontiersin.org/articles/10.3389/fenvs.2022.785517 DOI=10.3389/fenvs.2022.785517

MAGNANI, F., GRACE, J. & BORGHETTI, M. 2002. Adjustment of tree structure in response to the environment under hydraulic constraints. Functional Ecology, 16, 385-393.

Maherali, Hafiz, William T. Pockman, Robert B. Jackson 2004. Adaptive Variation in the Vulnerability of Woody Plants to Xylem Cavitation Ecology, Vol. 85, No. 8 (Aug., 2004), pp. 2184-2199

Malik, A. A., & Bouskill, N. J. (2022). Drought impacts on microbial trait distribution and feedback to soil carbon cycling. Functional Ecology, 36, 1442– 1456. https://doi.org/10.1111/1365-2435.14010

MAIRE, V., MARTRE, P., KATTGE, J., GASTAL, F., ESSER, G., FONTAINE, S. & SOUSSANA, J.-F. 2012. The coordination of leaf photosynthesis links C and N fluxes in C3 plant species. PloS one, 7, e38345.

MARTIN-STPAUL, N. K., LIMOUSIN, J.-M., VOGT-SCHILB, H., RODRÍGUEZ-CALCERRADA, J., RAMBAL, S., LONGEPIERRE, D. & MISSON, L. 2013. The temporal response to drought in a Mediterranean evergreen tree: comparing a regional precipitation gradient and a throughfall exclusion experiment. Global Change Biology, 19, 2413-2426.

Manavalan LP, Guttikonda SK, Phan Tran LS, Nguyen HT (2009) Physiological and molecular approaches to improve drought resistance in soybean. Plant Cell Physiol 50:1260–1276

Martinez-Meier, A., Sanchez, L., Pastorino, M., Gallo, L. and Rozenberg, P., 2008. What is hot in tree rings? The wood density of surviving Douglas-firs to the 2003 drought and heat wave. Forest Ecology and Management, 256(4), pp.837-843.

Matos, Ilaíne.S., Bernardo M. Flores, Marina Hirota, Bruno H. P. Rosado Critical transitions in rainfall manipulation experiments on grasslands Ecology and evolution DOI: 10.1002/ece3.6072

MCDOWELL, N., POCKMAN, W. T., ALLEN, C. D., BRESHEARS, D. D., COBB, N., KOLB, T., PLAUT, J., SPERRY, J., WEST, A., WILLIAMS, D. G. & YEPEZ, E. A. 2008. Mechanisms of plant survival and mortality during drought: why do some plants survive while others succumb to drought? New Phytologist, 178, 719-739.

MERCADO, L. M., MEDLYN, B. E., HUNTINGFORD, C., OLIVER, R. J., CLARK, D. B., SITCH, S., ZELAZOWSKI, P., KATTGE, J., HARPER, A. B. & COX, P. M. 2018. Large sensitivity in land carbon storage due to geographical and temporal variation in the thermal response of photosynthetic capacity. New Phytologist, 218, 1462-1477.

MITCHARD, E. T. A. 2018. The tropical forest carbon cycle and climate change. Nature, 559, 527-534.

Niu X., Song L., Xiao Y., Ge W. Drought-tolerant plant growth-promoting rhizobacteria associated with foxtail millet in a semi-arid agroecosystem and their potential in alleviating drought stress. Front. Microbiol. 2018;8:2580. doi: 10.3389/fmicb.2017.02580

OLIVER, R. J., MERCADO, L. M., CLARK, D. B., HUNTINGFORD, C., TAYLOR, C. M., VIDALE, P. L., MCGUIRE, P. C., TODT, M., FOLWELL, S., SHAMSUDHEEN SEMEENA, V. & MEDLYN, B. E. 2022. Improved representation of plant physiology in the JULES-vn5.6 land surface model: photosynthesis, stomatal conductance and thermal acclimation. Geosci. Model Dev., 15, 5567-5592.

Olson, M. E., Soriano, D., Rosell, J. A., Anfodillo, T., Donoghue, M. J., Edwards, E. J., León-Gómez, C., Dawson, T., Camarero Martínez, J. J., Castorena, M., Echeverría, A., Espinosa, C. I., Fajardo, A., Gazol, A., Isnard, S., Lima, R. S., Marcati, C. R., & Méndez-Alonzo, R. (2018). Plant height and hydraulic vulnerability to drought and cold. Proceedings of the National Academy of Sciences of the United States of America, 115, 7551–7556.

Ovenden, T.S., Perks, M.P., Clarke, T., Mencuccini, M., Jump, A.S., 2021b. Life after recovery: Increased resolution of forest resilience assessment sheds new light on post-drought compensatory growth and recovery dynamics. J. Ecol. 1–14.

PAN, Y., BIRDSEY, R. A., FANG, J., HOUGHTON, R., KAUPPI, P. E., KURZ, W. A., PHILLIPS, O. L., SHVIDENKO, A., LEWIS, S. L., CANADELL, J. G., CIAIS, P., JACKSON, R. B., PACALA, S. W., MCGUIRE, A. D., PIAO, S., RAUTIAINEN, A., SITCH, S. & HAYES, D. 2011. A Large and Persistent Carbon Sink in the World’s Forests. Science, 333, 988-993.

PATRA, P. K., CRISP, D., KAISER, J. W., WUNCH, D., SAEKI, T., ICHII, K., SEKIYA, T., WENNBERG, P. O., FEIST, D. G., POLLARD, D. F., GRIFFITH, D. W. T., VELAZCO, V. A., DE MAZIERE, M., SHA, M. K., ROEHL, C., CHATTERJEE, A. & ISHIJIMA, K. 2017. The Orbiting Carbon Observatory (OCO-2) tracks 2–3 peta-gram increase in carbon release to the atmosphere during the 2014–2016 El Niño. Scientific Reports, 7, 13567.

PENG, Y., BLOOMFIELD, K. J., CERNUSAK, L. A., DOMINGUES, T. F. & COLIN PRENTICE, I. 2021. Global climate and nutrient controls of photosynthetic capacity. Commun. Biol., 4, 462.

Petr, M., Boerboom, L.G., van der Veen, A. and Ray, D., 2014. A spatial and temporal drought risk assessment of three major tree species in Britain using probabilistic climate change projections. Climatic change, 124(4), pp.791-803.

PRENTICE, I. C., DONG, N., GLEASON, S. M., MAIRE, V. & WRIGHT, I. J. 2014. Balancing the costs of carbon gain and water transport: testing a new theoretical framework for plant functional ecology. Ecology letters, 17, 82-91.

QIE, L., LEWIS, S. L., SULLIVAN, M. J. P., LOPEZ-GONZALEZ, G., PICKAVANCE, G. C., SUNDERLAND, T., ASHTON, P., HUBAU, W., ABU SALIM, K., AIBA, S.-I., BANIN, L. F., BERRY, N., BREARLEY, F. Q., BURSLEM, D. F. R. P., DANČÁK, M., DAVIES, S. J., FREDRIKSSON, G., HAMER, K. C., HÉDL, R., KHO, L. K., KITAYAMA, K., KRISNAWATI, H., LHOTA, S., MALHI, Y., MAYCOCK, C., METALI, F., MIRMANTO, E., NAGY, L., NILUS, R., ONG, R., PENDRY, C. A., POULSEN, A. D., PRIMACK, R. B., RUTISHAUSER, E., SAMSOEDIN, I., SARAGIH, B., SIST, P., SLIK, J. W. F., SUKRI, R. S., SVÁTEK, M., TAN, S., TJOA, A., VAN NIEUWSTADT, M., VERNIMMEN, R. R. E., YASSIR, I., KIDD, P. S., FITRIADI, M., IDERIS, N. K. H., SERUDIN, R. M., ABDULLAH LIM, L. S., SAPARUDIN, M. S. & PHILLIPS, O. L. 2017. Long-term carbon sink in Borneo's forests halted by drought and vulnerable to edge effects. Nature Communications, 8, 1966.

Ray, D., Bathgate, S., Moseley, D., Taylor, P., Nicoll, B., Pizzirani, S. and Gardiner, B., 2015. Comparing the provision of ecosystem services in plantation forests under alternative climate change adaptation management options in Wales. Regional Environmental Change, 15(8), pp.1501-1513.

Read, J. and Sanson, G.D. (2003), Characterizing sclerophylly: the mechanical properties of a diverse range of leaf types. New Phytologist, 160: 81-99. https://doi.org/10.1046/j.1469-8137.2003.00855.x

Resende M, Munoz P, Acosta J et al., (2012) Accelerating the domestication of trees using genomic selection: accuracy of prediction models across ages and environments. New Phytol 193:617–624

ROGERS, A., MEDLYN, B. E., DUKES, J. S., BONAN, G., VON CAEMMERER, S., DIETZE, M. C., KATTGE, J., LEAKEY, A. D. B., MERCADO, L. M., NIINEMETS, Ü., PRENTICE, I. C., SERBIN, S. P., SITCH, S., WAY, D. A. & ZAEHLE, S. 2017. A roadmap for improving the representation of photosynthesis in Earth system models. New Phytol, 213, 22-42.

Roland Hasibeder, Lucia Fuchslueger, Andreas Richter and Michael Bahn (2015) Summer drought alters carbon allocation to roots and root respiration in mountain grassland New Phytologist (2015) 205: 1117–1127 doi: 10.1111/nph.13146

Rubin, R.L., van Groenigen, K.J. & Hungate, B.A. Plant growth promoting rhizobacteria are more effective under drought: a meta-analysis. Plant Soil 416, 309–323 (2017).

SCHLESINGER, W. H. & JASECHKO, S. 2014. Transpiration in the global water cycle. Agricultural and Forest Meteorology, 189-190, 115-117.

SCHWAAB, J., MEIER, R., MUSSETTI, G., SENEVIRATNE, S., BÜRGI, C. & DAVIN, E. L. 2021. The role of urban trees in reducing land surface temperatures in European cities. Nature Communications, 12, 6763.

Seleiman MF, Al-Suhaibani N, Ali N, Akmal M, Alotaibi M, Refay Y, Dindaroglu T, Abdul-Wajid HH, Battaglia ML. Drought Stress Impacts on Plants and Different Approaches to Alleviate Its Adverse Effects. Plants (Basel). 2021 Jan 28;10(2):259. doi: 10.3390/plants10020259. PMID: 33525688; PMCID: PMC7911879.

Sheffield J, Wood EF (2011). Drought: past problems and future scenarios. Earthscan, London, p 210 ISBN 978-184971-082-4

Shinwari, Z.K., Jan, S.A., Nakashima, K. et al., Genetic engineering approaches to understanding drought tolerance in plants. Plant Biotechnol Rep 14, 151–162 (2020). https://doi.org/10.1007/s11816-020-00598-6

Simard S, Asay A, Beiler K, Bingham M, Deslippe J, He X, Teste F (2015) Resource transfer between plants through ectomycorrhizal fungal networks. Mycorrhizal networks. Springer, Dordrecht, pp 133–176

SITCH, S., FRIEDLINGSTEIN, P., GRUBER, N., JONES, S. D., MURRAY-TORTAROLO, G., AHLSTRÖM, A., DONEY, S. C., GRAVEN, H., HEINZE, C., HUNTINGFORD, C., LEVIS, S., LEVY, P. E., LOMAS, M., POULTER, B., VIOVY, N., ZAEHLE, S., ZENG, N., ARNETH, A., BONAN, G., BOPP, L., CANADELL, J. G., CHEVALLIER, F., CIAIS, P., ELLIS, R., GLOOR, M., PEYLIN, P., PIAO, S. L., LE QUÉRÉ, C., SMITH, B., ZHU, Z. & MYNENI, R. 2015. Recent trends and drivers of regional sources and sinks of carbon dioxide. Biogeosciences, 12, 653-679.

SLOT, M., RIFAI, S. W. & WINTER, K. 2021. Photosynthetic plasticity of a tropical tree species, Tabebuia rosea, in response to elevated temperature and CO2. Plant, Cell & Environment, 2021;1–18 . DOI: 10.1111/pce.14049

SMITH, N. G. & KEENAN, T. F. 2020. Mechanisms underlying leaf photosynthetic acclimation to warming and elevated CO2 as inferred from least-cost optimality theory. GCB, 26, 5202-5216.

SMITH, N. G., KEENAN, T. F., COLIN PRENTICE, I., WANG, H., WRIGHT, I. J., NIINEMETS, Ü., CROUS, K. Y., DOMINGUES, T. F., GUERRIERI, R. & YOKO ISHIDA, F. 2019. Global photosynthetic capacity is optimized to the environment. Ecol Lett, 22, 506-517.

SMITH, N. G., MALYSHEV, S. L., SHEVLIAKOVA, E., KATTGE, J. & DUKES, J. S. 2016. Foliar temperature acclimation reduces simulated carbon sensitivity to climate. Nature Climate Change, 6, 407-411.

Sondergaard TE, Schulz A, Palmgren MG. Energization of transport processes in plants. roles of the plasma membrane H+-ATPase. Plant Physiol. 2004 Sep;136(1):2475-82. doi: 10.1104/pp.104.048231. PMID: 15375204; PMCID: PMC523315.

Sohn, J.A., Saha, S. and Bauhus, J., 2016. Potential of forest thinning to mitigate drought stress: A meta-analysis. Forest Ecology and Management, 380, pp.261-273.

Stokes, V., Kerr, G. and Connolly, T., 2020. Underplanting is a practical silvicultural method for regenerating and diversifying conifer stands in Britain. Forestry: An International Journal of Forest Research. DOI:10.1093/forestry/cpaa027

Takahashi Fuminori, Takashi Kuromori, Kaoru Urano, Kazuko Yamaguchi-Shinozaki and Kazuo Shinozaki (2020) Drought Stress Responses and Resistance in Plants: From Cellular Responses to Long-Distance Intercellular Communication. Frontiers in plan Science Volume 11 Article 556972. doi: 10.3389/fpls.2020.556972

Tangney, R., Paroissien, R., Le Breton, T.D. et al., Success of post-fire plant recovery strategies varies with shifting fire seasonality. Commun Earth Environ 3, 126 (2022). https://doi.org/10.1038/s43247-022-00453-2

THARAMMAL, T., BALA, G., DEVARAJU, N. & NEMANI, R. 2019. A review of the major drivers of the terrestrial carbon uptake: model-based assessments, consensus, and uncertainties. Environmental Research Letters, 14, 093005.

Thomas (2000). Vertical rooting patterns of mature Quercus trees growing on different soil types in northern Germany. Plant Ecology 147: 95–103

Thrum, Eric, A.,Enno Uhl, Hans Pretzsch Mixture reduces climate sensitivity of douglas fir stem growth Forest Ecol and Mangemeent (2016) 376 205-220

Ullah A, Akbar A, Luo Q, Khan AH, Manghwar H, Shaban M, Yang X (2019a) Microbiome diversity in cotton rhizosphere under normal and drought conditions. Microb Ecol 77:429–439

Ullah A, Nisar M, Ali H, Hazrat A, Hayat K, Keerio AA, Khan A (2019b) Drought tolerance improvement in plants: an endophytic bacterial approach. Appl Microbiol Biotechnol 103:7385–7397

Van Dyke, Mary N., Jonathan M. Levine & Nathan J. B. Kraft. 2022 Small rainfall changes drive substantial changes in plant coexistence Nature 611 p 507-523 doi.org/10.1038/s41586-022-05391-9

Varshney, RK, Barmukh, R, Roorkiwal, M, et al., Breeding custom-designed crops for improved drought adaptation. Advanced Genetics. 2021; 2(3):e202100017. doi:10.1002/ggn2.202100017

VERHOEF, A. & EGEA, G. 2014. Modeling plant transpiration under limited soil water: Comparison of different plant and soil hydraulic parameterizations and preliminary implications for their use in land surface models. Agricultural and Forest Meteorology, 191, 22-32.

VIDALE, P. L., EGEA, G., MCGUIRE, P. C., TODT, M., PETERS, W., MÜLLER, O., BALAN-SAROJINI, B. & VERHOEF, A. 2021. On the Treatment of Soil Water Stress in GCM Simulations of Vegetation Physiology. Frontiers in Environmental Science, 9.

Vogel A, Scherer-Lorenzen M, Weigelt A (2012) Grassland Resistance and Resilience after Drought Depends on Management Intensity and Species Richness. PLoS ONE 7(5): e36992. doi:10.1371/journal.pone.0036992

WALKER, A. P., DE KAUWE, M. G., BASTOS, A., BELMECHERI, S., GEORGIOU, K., KEELING, R. F., MCMAHON, S. M., MEDLYN, B. E., MOORE, D. J. P., NORBY, R. J., ZAEHLE, S., ANDERSON-TEIXEIRA, K.

WANG, H., PRENTICE, I. C., KEENAN, T. F., DAVIS, T. W., WRIGHT, I. J., CORNWELL, W. K., EVANS, B. J. & PENG, C. 2017. Towards a universal model for carbon dioxide uptake by plants. Nat Plants, 3, 734-741.

Yamori, W., Hikosaka, K., & Way, D. A. (2014). Temperature response of photosynthesis in C3, C4, and CAM plants: Temperature acclimation and temperature adaptation. Photosynthesis Research, 119(1), 101–117.

WAY, D. A. & YAMORI, W. 2014. Thermal acclimation of photosynthesis: on the importance of adjusting our definitions and accounting for thermal acclimation of respiration. Photosynthesis Research, 119, 89-100.

WILTSHIRE, R. A. J. & ZAEHLE, S. 2019. Global Carbon Budget 2019. Earth Syst. Sci. Data, 11, 1783-1838.

WINBOURNE, J. B., JONES, T. S., GARVEY, S. M., HARRISON, J. L., WANG, L., LI, D., TEMPLER, P. H. & HUTYRA, L. R. 2020. Tree Transpiration and Urban Temperatures: Current Understanding, Implications, and Future Research Directions. BioScience, 70, 576-588.

WINKLER, A. J., MYNENI, R. B., HANNART, A., SITCH, S., HAVERD, V., LOMBARDOZZI, D., ARORA, V. K., PONGRATZ, J., NABEL, J. E. M. S., GOLL, D. S., KATO, E., TIAN, H., ARNETH, A., FRIEDLINGSTEIN, P., JAIN, A. K., ZAEHLE, S. & BROVKIN, V. 2021. Slowdown of the greening trend in natural vegetation with further rise in atmospheric CO2. Biogeosciences, 18, 4985-5010.

Wu, Q.S.; Zou, Y.N. Arbuscular mycorrhizal fungi and tolerance of drought stress in plants. In Arbuscular Mycorrhizas and Stress Tolerance of Plants; Springer: Singapore, 2017; pp. 25–41.

Xi, Nianxun Chen, Dongxia Bahn, Michael Wu, Hangyu Chu, Chengjin Cadotte, Marc W. Bloor, Juliette M. G. (2022) Drought soil legacy alters drivers of plant diversity-productivity relationships in oldfield systems. Science Advances 2022 Vol. 8 Issue (18) DOI: doi:10.1126/sciadv.abn3368

XIA, J., YUAN, W., WANG, Y.-P. & ZHANG, Q. 2017. Adaptive Carbon Allocation by Plants Enhances the Terrestrial Carbon Sink. Scientific Reports 3341

Xu, Q.; Fu, H.; Zhu, B.; Hussain, H.A.; Zhang, K.; Tian, X.; Duan, M.; Xie, X.; Wang, L. Potassium Improves Drought Stress Tolerance in Plants by Affecting Root Morphology, Root Exudates, and Microbial Diversity. Metabolites 2021, 11, 131. https://doi.org/10.3390/metabo11030131

YANG, Y., SAATCHI, S. S., XU, L., YU, Y., CHOI, S., PHILLIPS, N., KENNEDY, R., KELLER, M., KNYAZIKHIN, Y. & MYNENI, R. B. 2018. Post-drought decline of the Amazon carbon sink. Nature Communications, 9, 3172.

YORDANOV, I., VELIKOVA, V. & TSONEV, T. 2000. Plant Responses to Drought, Acclimation, and Stress Tolerance. Photosynthetica, 38, 171-186.

Zas R, Sampedro L (2015) Heritability of seed weight in maritime pine, a relevant trait in the transmission of environmental maternal effects. Heredity 114:116

ZENG, Z., PIAO, S., LI, L. Z. X., ZHOU, L., CIAIS, P., WANG, T., LI, Y., LIAN, X., WOOD, E. F., FRIEDLINGSTEIN, P., MAO, J., ESTES, L. D., MYNENI, RANGA B., PENG, S., SHI, X., SENEVIRATNE, S. I. & WANG, Y. 2017. Climate mitigation from vegetation biophysical feedbacks during the past three decades. Nature Climate Change, 7, 432-436.

ZHOU, S. X., MEDLYN, B. E. & PRENTICE, I. C. 2016. Long-term water stress leads to acclimation of drought sensitivity of photosynthetic capacity in xeric but not riparian Eucalyptus species. Ann Bot, 117, 133-44.

References: Section 2 Drought and wildfires impact on vegetation.

Abatzoglou, J. T. and Williams, A. P.: Impact of anthropogenic climate change on wildfire across western US forests, Proc. Natl. Acad. Sci. U. S. A., 113, 11770–11775, 2016.

Abatzoglou, J. T., Williams, A. P., and Barbero, R.: Global emergence of anthropogenic climate change in fire weather indices, Geophys. Res. Lett., 46, 326–336, 2019.

Abolafia-Rosenzweig, R., C. He and F. Chen. Winter and spring climate explains a large portion of interannual variability and trend in western U.S. summer fire burned area

Environmental Research Letters 2022 Vol. 17 Issue 5 Pages 054030. DOI: 10.1088/1748-9326/ac6886

Adams, M.A., Shadmanroodposhti, M. and Neumann, M. (2020), Causes and consequences of Eastern Australia's 2019–20 season of mega-fires: A broader perspective. Glob Change Biol, 26: 3756-3758. https://doi.org/10.1111/gcb.15125

Alencar, A. A., Brando, P. M., Asner, G. P., and Putz, F. E.: Landscape fragmentation, severe drought, and the new Amazon forest fire regime, Ecol. Appl., 25, 1493–1505, 2015.

Alvarado, S. T., Andela, N., Silva, T. S. F., and Archibald, S.: Thresholds of fire response to moisture and fuel load differ between tropical savannas and grasslands across continents, Glob. Ecol. Biogeogr., 29, 331–344, 2020

Andela, N., Morton, D. C., Giglio, L., Chen, Y., van der Werf, G. R., Kasibhatla, P. S., DeFries, R. S., Collatz, G. J., Hantson, S., Kloster, S., Bachelet, D., Forrest, M., Lasslop, G., Li, F., Mangeon, S., Melton, J. R., Yue, C., and Randerson, J. T.: A human-driven decline in global burned area, Science, 356, 1356–1362, 2017.

Balch, J. K., Nepstad, D. C., and Curran, L. M.: Pattern and process: Fire-initiated grass invasion at Amazon transitional forest edges, in: Tropical Fire Ecology: Climate Change, Land Use, and Ecosystem Dynamics, edited by: Cochrane, M. A., Springer Berlin Heidelberg, Berlin, Heidelberg, 481–502, 2009.

Boer, M. M., de Dios, V. R., and Bradstock, R. A.: Unprecedented burn area of Australian mega forest fires, Nat. Clim. Chang., 10, 171–172, 2020.

Borchers Arriagada, N., Palmer, A. J., Bowman, D. M., Morgan, G. G., Jalaludin, B. B., and Johnston, F. H.: Unprecedented smoke-related health burden associated with the 2019-20 bushfires in eastern Australia, Med. J. Aust., 213, 282–283, 2020.

Brady M.K., Dickinson M.B., Miesel J.R., Wonkka C.L., Kavanagh K.L., Lodge A.G., Rogers W.E., Starns H.D., Tolleson D.R., Treadwell M.L., Twidwell D. & Hanan E.J. 2022. Soil heating in fire (SheFire): a model and measurement method for estimating soil heating and effects during wildland fires. Ecological Applications 32, e2627. https://doi.org/10.1002/eap.2627

Dale, V.H., Joyce, L.A., McNulty, S., Neilson, R.P., Ayres, M.P., Flannigan, M.D. et al. (2001). Climate change and forest disturbances: climate change can affect forests by altering the frequency, intensity, duration, and timing of fire, drought, introduced species, insect and pathogen outbreaks, hurricanes, windstorms, ice storms, or landslides. BioScience, Volume 51, Issue 9, September 2001, Pages 723–734, https://doi.org/10.1641/0006-3568(2001)051[0723:CCAFD]2.0.CO;2

Clements, Craig B., Neil P. Lareau, Daisuke Seto, Jonathan Contezac, Braniff Davis Casey Teske, Thomas J. Zajkowski, Andrew T. Hudak, Benjamin C. Bright, Matthew B. Dickinson, Bret W. Butler, Daniel Jimenez and J. Kevin Hiers. Fire weather conditions and fire–atmosphere interactions observed during low-intensity prescribed fires – RxCADRE 2012. International Journal of Wildland Fire 25(1) 90-101 https://doi.org/10.1071/WF14173, 2015

Flannigan, M. D., Wotton, B. M., Marshall, G. A., de Groot, W. J., Johnston, J., Jurko, N., and Cantin, A. S.: Fuel moisture sensitivity to temperature and precipitation: climate change implications, Clim. Change, 134, 59–71, 2016.

Finney, M., Grenfell, I.C. and McHugh, C.W. (2009). Modeling containment of large wildfires using generalized linear mixed-model analysis. Forest Science, 55(3), 249-255.

Forkel, M., Dorigo, W., Lasslop, G., Teubner, I., Chuvieco, E., and Thonicke, K.: A data-driven approach to identify controls on global fire activity from satellite and climate observations (SOFIA V1), Geoscientific Model Development, 10, 4443–4476, 2017.

Geirinhas, J.L., Trigo, R.M., Libonati, R., Coelho, C.A. and Palmeira, A.C. (2018). Climatic and synoptic characterization of heat waves in Brazil. International Journal of Climatology, 38(4), 1760-1776. https://doi.org/10.1002/joc.5294.

Haas, O., Prentice, I. C., and Harrison, S. P.: Global environmental controls on wildfire burnt area, size, and intensity, Environ. Res. Lett., 17, 065004, 2022.

Hantson, S., Arneth, A., Harrison, S. P., Kelley, D. I., Prentice, I. C., Rabin, S. S., Archibald, S., Mouillot, F., Arnold, S. R., Artaxo, P., Bachelet, D., Ciais, P., Forrest, M., Friedlingstein, P., Hickler, T., Kaplan, J. O., Kloster, S., Knorr, W., Laslop, G., Li, F., Melton, J. R., Meyn, A., Sitch, S., Spessa, A., van der Werf, G. R., Voulgarakis, A., and Yue, C.: The status and challenge of global fire modelling, Biogeosciences, 13, 3359–3375, 2016.

Hantson, S., Kelley, D. I., Arneth, A., Harrison, S. P., Archibald, S., Bachelet, D., Forrest, M., Hickler, T., Lasslop, G., Li, F., Mangeon, S., Melton, J. R., Nieradzik, L., Rabin, S. S., Colin Prentice, I., Sheehan, T., Sitch, S., Teckentrup, L., Voulgarakis, A., and Yue, C.: Quantitative assessment of fire and vegetation properties in simulations with fire-enabled vegetation models from the Fire Model Intercomparison Project, https://doi.org/10.5194/gmd-13-3299-2020, 2020.

Head, L., Adams, M., and McGregor, H. V.: Climate change and Australia, Wiley Interdisciplinary, 2014.

Holloway, J. E. and Lewkowicz, A. G.: Impact of wildfire on permafrost landscapes: A review of recent advances and future prospects, Permafrost Periglacial Processes, 2020.

Johnston, F. H., Henderson, S. B., Chen, Y., Randerson, J. T., Marlier, M., Defries, R. S., Kinney, P., Bowman, D. M. J. S., and Brauer, M.: Estimated global mortality attributable to smoke from landscape fires, Environ. Health Perspect., 120, 695–701, 2012.

Jolly, W. M., Cochrane, M. A., Freeborn, P. H., Holden, Z. A., Brown, T. J., Williamson, G. J., and Bowman, D. M. J. S.: Climate-induced variations in global wildfire danger from 1979 to 2013, Nat. Commun., 6, 7537, 2015.

Jones, M. W., Abatzoglou, J. T., Veraverbeke, S., Andela, N., Lasslop, G., Forkel, M., Smith, A. J. P., Burton, C., Betts, R. A., van der Werf, G. R., Sitch, S., Canadell, J. G., Santín, C., Kolden, C., Doerr, S. H., and Le Quéré, C.: Global and regional trends and drivers of fire under climate change, Rev. Geophys., 60, https://doi.org/10.1029/2020rg000726, 2022.

Kelley, D. I., Bistinas, I., Whitley, R., Burton, C., Marthews, T. R., and Dong, N.: How contemporary bioclimatic and human controls change global fire regimes, Nat. Clim. Chang., 9, 690–696, 2019.

Kelley, D. I., Burton, C., Huntingford, C., Brown, M. A. J., Whitley, R., and Dong, N.: Low meteorological influence found in 2019 Amazonia fires, Biogeosciences, 18, 787–804, 2021.

Knorr, W., Jiang, L., and Arneth, A.: Climate, CO2 and human population impacts on global wildfire emissions, Biogeosciences, 13, 267–282, 2016.

Krawchuk, M.A. and Moritz, M.A. (2011). Constraints on global fire activity vary across a resource gradient. Ecology, 92(1), 121-132. https://doi.org/10.1890/09-1843.1

Littell, J. S., Peterson, D. L., Riley, K. L., Liu, Y., and Luce, C. H.: A review of the relationships between drought and forest fire in the United States, Glob. Chang. Biol., 22, 2353–2369, 2016.

Mansoor,S., I. Farooq, M. M. Kachroo, A. E. D. Mahmoud, M. Fawzy, S. M., Popescu, M.N Alyemeni, C. Sonne, J. Rinklebe, P. Ahmad, (2022) Elevation in wildfire frequencies with respect to the climate change. Journal of Environmental Management 2022 Vol. 301 Pages 113769. https://doi.org/10.1016/j.jenvman.2021.113769

McCarty, J. L., Smith, T. E. L., and Turetsky, M. R.: Arctic fires re-emerging, Nat. Geosci., 13, 658–660, 2020.

McKenzie, D. and Littell, J.S. (2017). Climate change and the eco-hydrology of fire: will area burned increase in a warming western USA? Ecological Applications, 27(1), 26-36. https://doi.org/10.1002/eap.1420

van Oldenborgh, G. J., Krikken, F., Lewis, S., Leach, N. J., Lehner, F., Saunders, K. R., van Weele, M., Haustein, K., Li, S., Wallom, D., Sparrow, S., Arrighi, J., Singh, R. P., van Aalst, M. K., Philip, S. Y., Vautard, R., and Otto, F. E. L.: Attribution of the Australian bushfire risk to anthropogenic climate change, Natural Hazards and Earth System Sciences Discussions, https://doi.org/10.5194/nhess-2020-69, 2020.

Osterkamp, T. E.: The recent warming of permafrost in Alaska, Glob. Planet. Change, 49, 187–202, 2005.

Panisset, J.S., Libonati, R., Gouveia, C.M.P., Machado-Silva, F., Franca, D.A., Franca, J.R.A. et al. (2018). Contrasting patterns of the extreme drought episodes of 2005, 2010 and 2015 in the Amazon Basin. International Journal of Climatology, 38(2), 1096-1104. https://doi.org/10.1002/joc.5224

Prentice, I. C., Kelley, D. I., and Foster, P. N.: Modeling fire and the terrestrial carbon balance, Global, 2011.

Rabin, Sam S. Joe R. Melton , Gitta Lasslop , Dominique Bachelet , Matthew Forrest , Stijn Hantson , Jed O. Kaplan , Fang Li , Stéphane Mangeon, Daniel S. Ward, Chao Yue, Vivek K. Arora, Thomas Hickler , Silvia Kloster , Wolfgang Knorr, Lars Nieradzik, Allan Spessa, Gerd A. Folberth, Tim Sheehan6 , Apostolos Voulgarakis, Douglas I. Kelley, I. Colin Prentice, Stephen Sitch, Sandy Harrison, and Almut Arneth (2014). The Fire Modeling Intercomparison Project (FireMIP), phase 1: experimental and analytical protocols with detailed model descriptions Geosci. Model Dev., 10, 1175–1197, 2017 www.geosci-model-dev.net/10/1175/2017/ doi:10.5194/gmd-10-1175-2017

Restaino, C.M. and Safford, H.D. (2018). Fire and climate change. In Fire in California's Ecosystems. Van Wagtendonk, J.W., Sugihara, N.G., Stephens, S.L., Thode, A.E., Shaffer, K.E. and Fites-Kaufman, J. (eds.). Second edition. Berkeley

Sharples, J. J., Cary, G. J., Fox-Hughes, P., Mooney, S., Evans, J. P., Fletcher, M.-S., Fromm, M., Grierson, P. F., McRae, R., and Baker, P.: Natural hazards in Australia: extreme bushfire, Clim. Change, 139, 85–99, 2016.

da Silva Junior, C. A., Teodoro, P. E., Delgado, R. C., Teodoro, L. P. R., Lima, M., de Andréa Pantaleão, A., Baio, F. H. R., de Azevedo, G. B., de Oliveira Sousa Azevedo, G. T., Capristo-Silva, G. F., Arvor, D., and Facco, C. U.: Persistent fire foci in all biomes undermine the Paris Agreement in Brazil, Sci. Rep., 10, 16246, 2020.

Swanson, D. K., Sousanes, P. J., and Hill, K.: Increased mean annual temperatures in 2014–2019 indicate permafrost thaw in Alaskan national parks, Arct. Antarct. Alp. Res., 53, 1–19, 2021.

UNEP, Popescu, A., Paulson, A. K., Christianson, A. C., Sullivan, A., Tulloch, A., Bilbao, B., Mathison, C., Robinson, C., Burton, C., Ganz, D., Nangoma, D., Saah, D., Armenteras, D., Driscoll, D., Hankins, D. L., Kelley, D. I., Langer, E. R. L., Baker, E., Berenguer, E., Reisen, F., Robinne, F.-N., Galudra, G., Humphrey, G., Safford, H., Baird, I. G., Oliveras, I., Littell, J., Kieft, J., Chew, J., Maclean, K., Wittenberg, L., Anderson, L. O., Gillson, L., Plucinski, M., Moritz, M., Brown, M., Soto, M. C., Flannigan, M., Costello, O., Silva, P. S., Fernandes, P., Moore, P., Jandt, R., Blanchi, R., Libonati, R., Archibald, S., Dunlop, S., McCaffrey, S., Page, S., Gonzãƒâ¡lez, T. D. T., Sokchea, T., and Charlton, V.: Spreading like Wildfire: The Rising Threat of Extraordinary Landscape Fires, edited by: Sullivan, A., Kurvits, T., and E., B., United Nations Environment Programme and GRID-Arendal., 2022.

Ward, M., Tulloch, A. I. T., Radford, J. Q., Williams, B. A., Reside, A. E., Macdonald, S. L., Mayfield, H. J., Maron, M., Possingham, H. P., Vine, S. J., O'Connor, J. L., Massingham, E. J., Greenville, A. C., Woinarski, J. C. Z., Garnett, S. T., Lintermans, M., Scheele, B. C., Carwardine, J., Nimmo, D. G., Lindenmayer, D. B., Kooyman, R. M., Simmonds, J. S., Sonter, L. J., and Watson, J. E. M.: Impact of 2019–2020 mega-fires on Australian fauna habitat, Nature Ecology & Evolution, 4, 1321–1326, 2020.

Williams, A. P., Abatzoglou, J. T., Gershunov, A., Guzman-Morales, J., Bishop, D. A., Balch, J. K., and Lettenmaier, D. P.: Observed impacts of anthropogenic climate change on wildfire in California, Earths Future, 7, 892–910, 2019.

Yeo, C. S., Kepert, J. D., and Hicks, R.: Fire danger indices: current limitations and a pathway to better indices, Bushfire & Natural Hazards CRC, Melbourne, Victoria, Australia, 2015.

References Section: 3 Drought induced wildfire and the impact on water quality in the UK

Bladon, K. D., Emelko, M. B., Silins, U., & Stone, M. (2014). Wildfire and the Future of Water Supply. Environmental Science & Technology, 48(16), 8936–8943. https://doi.org/10.1021/es500130g

Bodí, Merche B., Deborah A. Martin Victoria N.Balfour Cristina Santín, StefanH. Doerr, Paulo Pereira, Artemi Cerdà, Jorge Mataix-Solera (2014) Wildland fire ash: Production, composition and eco-hydro-geomorphic effects. Earth-Science Reviews 130 (2014) 103–127

Charette, T., Prepas, E.E., 2003. Wildfire impacts on phytoplankton communities of three small lakes on the Boreal Plain, Alberta, Canada: a paleolimnological study. Can. J. Fish. Aquat. Sci. 60, 584–593.

Costa, M. R., Calvão, A. R., & Aranha, J. (2014). Linking wildfire effects on soil and water chemistry of the Marão River watershed, Portugal, and biomass changes detected from Landsat imagery. Applied Geochemistry, 44, 93–102. https://doi.org/10.1016/j.apgeochem.2013.09.009

Cowell, A.L., Matthews, T.G., Lind, P.R., 2006. Effect of fire on benthic algal assemblage structure and recolonization in intermittent streams. Austral Ecol. 31, 696–707

Earl, S.R., Blinn, D.W., 2003. Effects of wildfire ash on water chemistry and biota in southwestern U.S.A. streams. Freshw. Biol. 48, 1015–1030.

Evans, C. D., Malcolm, I. A., Shilland, E. M., Rose, N. L., Turner, S. D., Crilly, A., et al., (2017). Sustained Biogeochemical Impacts of Wildfire in a Mountain Lake Catchment. Ecosystems. https://doi.org/10.1007/s10021-016-0064-1

Fernandez-Garcia, P.; Anderson, R.; Felgate, S.; Gaffney, P.; Gilbert, P.; Grant, A.; Mayor, D.; Pickard, A.E. (2022). Water chemistry measured from peatland sub-catchments in the Flow Country, Scotland following a wildfire 2019-2020. NERC EDS Environmental Information Data Centre. https://doi.org/10.5285/57748e4f-d0a4-4648-8a61-bd1c2066db1e

Granath, G., Evans, C. D., Strengbom, J., Fölster, J., Grelle, A., Strömqvist, J., & Köhler, S. J. (2021). The impact of wildfire on biogeochemical fluxes and water quality in boreal catchments. Biogeosciences, 18(10), 3243–3261. https://doi.org/10.5194/bg-18-3243-2021

Hall, S.J., Lombardozzi, D., 2008. Short-term effects ofwildfire on montane stream ecosystems in the southern Rocky Mountains: one and two years post-burn. West. N. Am. Nat. 68, 453–462.

LYON, J. P., & O'CONNOR, J. P. (2008). Smoke on the water: Can riverine fish populations recover following a catastrophic fire-related sediment slug? Austral Ecology, 33(6), 794–806. https://doi.org/10.1111/j.1442-9993.2008.01851.x

Minshall, G. W., Brock, J. T., Andrews, D. A., & Robinson, C. T. (2001). Water quality, substratum and biotic responses of five central Idaho (USA) streams during the first year following the Mortar Creek fire. International Journal of Wildland Fire, 10(2), 185–199. Retrieved from https://doi.org/10.1071/WF01017

Nunes, Bruno, Vera Silva, Isabel Campos, Joana Luísa Pereira, Patrícia Pereira, Jan Jacob Keizer, Fernando Gonçalves, Nelson Abrantes, Off-site impacts of wildfires on aquatic systems — Biomarker responses of the mosquitofish Gambusia holbrooki, Science of The Total Environment, Volumes 581–582, 2017, Pages 305-313, ISSN 0048-9697, https://doi.org/10.1016/j.scitotenv.2016.12.129

Paul, M. J., LeDuc, S. D., Lassiter, M. G., Moorhead, L. C., Noyes, P. D., & Leibowitz, S. G. (2022). Wildfire Induces Changes in Receiving Waters: A Review With Considerations for Water Quality Management. Water Resources Research, 58(9), e2021WR030699. https://doi.org/10.1029/2021WR030699

PILKINGTON, M. G., CAPORN, S. J. M., CARROLL, J. A., CRESSWELL, N., PHOENIX, G. K., LEE, J. A., et al., (2007). Impacts of burning and increased nitrogen deposition on nitrogen pools and leaching in an upland moor. Journal of Ecology, 95(6), 1195–1207. https://doi.org/10.1111/j.1365-2745.2007.01292.x

Ramchunder, S. J., Brown, L. E., & Holden, J. (2013). Rotational vegetation burning effects on peatland stream ecosystems. Journal of Applied Ecology, 50(3), 636–648. https://doi.org/https://doi.org/10.1111/1365-2664.12082

Robinne, F.-N., Hallema, D. W., Bladon, K. D., Flannigan, M. D., Boisramé, G., Bréthaut, C. M., et al., (2021). Scientists' warning on extreme wildfire risks to water supply. Hydrological Processes, 35(5), e14086. https://doi.org/10.1002/hyp.14086

Spencer, C.N., Gabel, K.O., Hauer, F.R., 2003. Wildfire effects on stream food webs and nutrient dynamics in Glacier National Park, USA. For. Ecol. Manag. 178, 141–153.

Smith, H. G., Sheridan, G. J., Lane, P. N. J., Nyman, P., & Haydon, S. (2011). Wildfire effects on water quality in forest catchments: A review with implications for water supply. Journal of Hydrology, 396(1), 170–192. https://doi.org/10.1016/j.jhydrol.2010.10.043

Son, J.-H., Kim, S., & Carlson, K. H. (2015). Effects of Wildfire on River Water Quality and Riverbed Sediment Phosphorus. WATER AIR AND SOIL POLLUTION, 226(3). https://doi.org/10.1007/s11270-014-2269-2

Williamson, J., Evans, C., Spears, B., Pickard, A., Chapman, P. J., Feuchtmayr, H., et al., (2022). Understanding the impacts of peatland catchment management on DOM concentration and treatability. Biogeosciences Discussions, 2022, 1–24. https://doi.org/10.5194/bg-2022-209

References: Section 4 Detection of drought impacts on vegetation from Earth Observation data

Alahacoon, N. and M. Edirisinghe, A comprehensive assessment of remote sensing and traditional based drought monitoring indices at global and regional scale. Geomatics, Natural Hazards and Risk, 2022. 13(1): p. 762-799. DOI: 10.1080/19475705.2022.2044394.

Allen, C.D., A.K. Macalady, H. Chenchouni, D. Bachelet, N. McDowell, M. Vennetier, T. Kitzberger, A. Rigling, D.D. Breshears, E.H. Hogg, P. Gonzalez, R. Fensham, Z. Zhang, J. Castro, N. Demidova, J.-H. Lim, G. Allard, S.W. Running, A. Semerci and N. Cobb, A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management, 2010. 259(4): p. 660-684. DOI: https://doi.org/10.1016/j.foreco.2009.09.001

Anyamba, A. and C.J. Tucker, Historical perspectives on AVHRR NDVI and vegetation drought monitoring. B.D. Wardlow, M.C. Anderson, J.P. Verdin (Eds.), Remote Sensing of Drought: Innovative Monitoring Approaches, CRC Press, New York, United States of America, 2012: p. 23-51. DOI: 10.1201/b11863.

Ashraf, M.A., M.J. Maah and I. Yusoff, Introduction to Remote Sensing of Biomass, in Biomass and Remote Sensing of Biomass, A. Islam, Editor. 2011, IntechOpen: Rijeka. p. Ch. 8.

Bachmair, S., C. Svensson, J. Hannaford, L.J. Barker and K. Stahl, A quantitative analysis to objectively appraise drought indicators and model drought impacts. Hydrol. Earth Syst. Sci., 2016. 20(7): p. 2589-2609. DOI: 10.5194/hess-20-2589-2016.

Bachmair, S., M. Tanguy, J. Hannaford and K. Stahl, How well do meteorological indicators represent agricultural and forest drought across Europe? Environmental Research Letters, 2018. 13(3): p. 034042. DOI: 10.1088/1748-9326/aaafda.

Blum, A., J. Mayer and G. Gozlan, Infrared thermal sensing of plant canopies as a screening technique for dehydration avoidance in wheat. Field Crops Research, 1982. 5: p. 137-146. DOI: https://doi.org/10.1016/0378-4290(82)90014-4

Bolton, D.K. and M.A. Friedl, Forecasting crop yield using remotely sensed vegetation indices and crop phenology metrics. Agricultural and Forest Meteorology, 2013. 173: p. 74-84. DOI: https://doi.org/10.1016/j.agrformet.2013.01.007

Bouras, E.h., L. Jarlan, S. Er-Raki, R. Balaghi, A. Amazirh, B. Richard and S. Khabba, Cereal Yield Forecasting with Satellite Drought-Based Indices, Weather Data and Regional Climate Indices Using Machine Learning in Morocco. Remote Sensing, 2021. 13(16): p. 3101.

Brown, J.F., B.D. Wardlow, T. Tadesse, M.J. Hayes and B.C. Reed, The Vegetation Drought Response Index (VegDRI): A New Integrated Approach for Monitoring Drought Stress in Vegetation. GIScience & Remote Sensing, 2008. 45(1): p. 16-46. DOI: 10.2747/1548-1603.45.1.16.

Brust, C., J.S. Kimball, M.P. Maneta, K. Jencso and R.H. Reichle, DroughtCast: A Machine Learning Forecast of the United States Drought Monitor. Frontiers in Big Data, 2021. 4. DOI: 10.3389/fdata.2021.773478.

Buras, A., C. Schunk, C. Zeiträg, C. Herrmann, L. Kaiser, H. Lemme, C. Straub, S. Taeger, S. Gößwein, H.-J. Klemmt and A. Menzel, Are Scots pine forest edges particularly prone to drought-induced mortality? Environmental Research Letters, 2018. 13(2): p. 025001. DOI: 10.1088/1748-9326/aaa0b4.

Byer, S. and Y. Jin, Detecting Drought-Induced Tree Mortality in Sierra Nevada Forests with Time Series of Satellite Data. Remote Sensing, 2017. 9(9): p. 929.

Coelho, A.P., R.T. de Faria, F.T. Leal, J.d.A. Barbosa and D.L. Rosalen, Validation of white oat yield estimation models using vegetation indices. Basic areas, Bragantia 79 (2), 2020. DOI: 10.1590/1678-4499.20190387.

Dubey, S.K., A.S. Gavli, S.K. Yadav, S. Sehgal and S.S. Ray, Remote Sensing-Based Yield Forecasting for Sugarcane (Saccharum officinarum L.) Crop in India. Journal of the

Indian Society of Remote Sensing, 2018. 46(11): p. 1823-1833. DOI: 10.1007/s12524-018-0839-2.

Esfandabadi, H.S., M.G. Asl, Z.S. Esfandabadi, S. Gautam and M. Ranjbari, Drought assessment in paddy rice fields using remote sensing technology towards achieving food security and SDG2. British Food Journal, 2022. 124(12): p. 4219-4233. DOI: 10.1108/BFJ-08-2021-0872.

Feng, P., B. Wang, D.L. Liu, C. Waters, D. Xiao, L. Shi and Q. Yu, Dynamic wheat yield forecasts are improved by a hybrid approach using a biophysical model and machine learning technique. Agricultural and Forest Meteorology, 2020. 285-286: p. 107922. DOI: https://doi.org/10.1016/j.agrformet.2020.107922

Allen, C.D. and D.D. Breshears, Drought-induced shift of a forest–woodland ecotone: Rapid landscape response to climate variation. Proceedings of the National Academy of Sciences, 1998. 95(25): p. 14839-14842. DOI: doi:10.1073/pnas.95.25.14839.

Gao, B-c., NDWI—A normalized difference water index for remote sensing of vegetation liquid water from space. Remote Sensing of Environment, 1996. 58(3): p. 257-266. DOI: https://doi.org/10.1016/S0034-4257(96)00067-3

García-León, D., S. Contreras and J. Hunink, Comparison of meteorological and satellite-based drought indices as yield predictors of Spanish cereals. Agricultural Water Management, 2019. 213: p. 388-396. DOI: https://doi.org/10.1016/j.agwat.2018.10.030

Gilabert, M.A., A. Moreno, F. Maselli, B. Martínez, M. Chiesi, S. Sánchez-Ruiz, F.J. García-Haro, A. Pérez-Hoyos, M. Campos-Taberner, O. Pérez-Priego, P. Serrano-Ortiz and A. Carrara, Daily GPP estimates in Mediterranean ecosystems by combining remote sensing and meteorological data. ISPRS Journal of Photogrammetry and Remote Sensing, 2015. 102: p. 184-197. DOI: https://doi.org/10.1016/j.isprsjprs.2015.01.017

Goodwin, N.R., N.C. Coops, M.A. Wulder, S. Gillanders, T.A. Schroeder and T. Nelson, Estimation of insect infestation dynamics using a temporal sequence of Landsat data. Remote Sensing of Environment, 2008. 112(9): p. 3680-3689. DOI: https://doi.org/10.1016/j.rse.2008.05.005

Gouveia, C.M., R.M. Trigo, S. Beguería and S.M. Vicente-Serrano, Drought impacts on vegetation activity in the Mediterranean region: An assessment using remote sensing data and multi-scale drought indicators. Global and Planetary Change, 2017. 151: p. 15-27. DOI: https://doi.org/10.1016/j.gloplacha.2016.06.011

Graw, V., O. Dubovyk, M. Duguru, P. Heid, G. Ghazaryan, J.C. Villagrán de León, J. Post, J. Szarzynski, D. Tsegai and Y. Walz, Chapter 9 - Assessment, monitoring, and early warning of droughts: the potential for satellite remote sensing and beyond, in

Current Directions in Water Scarcity Research, E. Mapedza, et al., Editors. 2019, Elsevier. p. 115-131.

Huete, A., K. Didan, T. Miura, E.P. Rodriguez, X. Gao and L.G. Ferreira, Overview of the radiometric and biophysical performance of the MODIS vegetation indices. Remote Sensing of Environment, 2002. 83(1): p. 195-213. DOI: https://doi.org/10.1016/S0034-4257(02)00096-2

Huete, A.R., N. Restrepo-Coupe, P. Ratana, K. Didan, S.R. Saleska, K. Ichii, S. Panuthai and M. Gamo, Multiple site tower flux and remote sensing comparisons of tropical forest dynamics in Monsoon Asia. Agricultural and Forest Meteorology, 2008. 148(5): p. 748-760. DOI: https://doi.org/10.1016/j.agrformet.2008.01.012

Hunt, M.L., G.A. Blackburn, L. Carrasco, J.W. Redhead and C.S. Rowland, High resolution wheat yield mapping using Sentinel-2. Remote Sensing of Environment, 2019. 233: p. 111410. DOI: https://doi.org/10.1016/j.rse.2019.111410

Jain, S.K., R. Keshri, A. Goswami, A. Sarkar and A. Chaudhry, Identification of drought-vulnerable areas using NOAA AVHRR data. International Journal of Remote Sensing, 2009. 30(10): p. 2653-2668. DOI: 10.1080/01431160802555788

Jiao, W., L. Zhang, Q. Chang, D. Fu, Y. Cen and Q. Tong, Evaluating an Enhanced Vegetation Condition Index (VCI) Based on VIUPD for Drought Monitoring in the Continental United States. Remote Sensing, 2016. 8(3): p. 224.

Justice, C.O., L. Giglio, S. Korontzi, J. Owens, J.T. Morisette, D. Roy, J. Descloitres, S. Alleaume, F. Petitcolin and Y. Kaufman, The MODIS fire products. Remote Sensing of Environment, 2002. 83(1): p. 244-262. DOI: https://doi.org/10.1016/S0034-4257(02)00076-7

Kogan, F.N., Global Drought Watch from Space. Bulletin of the American Meteorological Society, 1997. 78(4): p. 621-636. DOI: 10.1175/1520-0477(1997)078<0621:GDWFS>2.0.CO;2.

Kogan, F., L. Salazar and L. Roytman, Forecasting crop production using satellite-based vegetation health indices in Kansas, USA. International Journal of Remote Sensing, 2012. 33(9): p. 2798-2814. DOI: 10.1080/01431161.2011.621464.

Kyratzis, A.C., D.P. Skarlatos, G.C. Menexes, V.F. Vamvakousis and A. Katsiotis, Assessment of Vegetation Indices Derived by UAV Imagery for Durum Wheat Phenotyping under a Water Limited and Heat Stressed Mediterranean Environment. Frontiers in Plant Science, 2017. 8. DOI: 10.3389/fpls.2017.01114.

Lambert, M.-J., P.C.S. Traoré, X. Blaes, P. Baret and P. Defourny, Estimating smallholder crops production at village level from Sentinel-2 time series in Mali's cotton belt. Remote Sensing of Environment, 2018. 216: p. 647-657. DOI: https://doi.org/10.1016/j.rse.2018.06.036

Liu, W.T. and F.N. Kogan, Monitoring regional drought using the Vegetation Condition Index. International Journal of Remote Sensing, 1996. 17(14): p. 2761-2782. DOI: 10.1080/01431169608949106.

Ma, Y., L. Ma, Q. Zhang, C. Huang, X. Yi, X. Chen, T. Hou, X. Lv and Z. Zhang, Cotton Yield Estimation Based on Vegetation Indices and Texture Features Derived From RGB Image. Frontiers in Plant Science, 2022. 13. DOI: 10.3389/fpls.2022.925986.

Marusig, D., F. Petruzzellis, M. Tomasella, R. Napolitano, A. Altobelli and A. Nardini, Correlation of Field-Measured and Remotely Sensed Plant Water Status as a Tool to Monitor the Risk of Drought-Induced Forest Decline. Forests, 2020. 11(1): p. 77.

Martens, B., D.G. Miralles, H. Lievens, R. van der Schalie, R.A.M. de Jeu, D. Fernández-Prieto, H.E. Beck, W.A. Dorigo and N.E.C. Verhoest, GLEAM v3: satellite-based land evaporation and root-zone soil moisture. Geosci. Model Dev., 2017. 10(5): p. 1903-1925. DOI: 10.5194/gmd-10-1903-2017.

Maselli, F., S. Romanelli, L. Bottai and G. Maracchi, Processing of GAC NDVI data for yield forecasting in the Sahelian region. International Journal of Remote Sensing, 2000. 21(18): p. 3509-3523. DOI: 10.1080/014311600750037525.

Millard, J.P., Jackson, R.D.,Goettelman, R.C.; Reginato, R.J.. Crop water-stress assessment using an airborne thermal scanner [used to measure the temperature of a wheat crop canopy in Phoenix, Arizona, promise for irrigation scheduling and crop yield predictions. Photogrammetric-Engineering-and-Remote-Sensing, v. 44(1) p. 77-85, 1978.

Miranda, A., A. Lara, A. Altamirano, C. Di Bella, M.E. González and J. Julio Camarero, Forest browning trends in response to drought in a highly threatened mediterranean landscape of South America. Ecological Indicators, 2020. 115: p. 106401. DOI: https://doi.org/10.1016/j.ecolind.2020.106401

Monteith, J.L., Solar Radiation and Productivity in Tropical Ecosystems. The Journal of Applied Ecology, 1972. 9(3): p. 747. DOI: 10.2307/2401901.

Moyer, W., N. Patil, E. Toffoli, N. Hadi, A. Guan and S. Goings, Drought Monitoring with Drones: A Hundred Fields at a Time. NASA Technical Memorandum, NASA/TM-20220014853. Accessed online (17/01/2023): https://ntrs.nasa.gov/api/citations/20220014853/downloads/20220014853_VIP-Interns_Report-Drought-Monitoring.pdf 2022.

Nicolai-Shaw, N., J. Zscheischler, M. Hirschi, L. Gudmundsson and S.I. Seneviratne, A drought event composite analysis using satellite remote-sensing based soil moisture. Remote Sensing of Environment, 2017. 203: p. 216-225. DOI: https://doi.org/10.1016/j.rse.2017.06.014

Orth, R. and G. Destouni, Drought reduces blue-water fluxes more strongly than green-water fluxes in Europe. Nature Communications, 2018. 9(1): p. 3602. DOI: 10.1038/s41467-018-06013-7.

Peng, C., Z. Ma, X. Lei, Q. Zhu, H. Chen, W. Wang, S. Liu, W. Li, X. Fang and X. Zhou, A drought-induced pervasive increase in tree mortality across Canada's boreal forests. Nature Climate Change, 2011. 1(9): p. 467-471. DOI: 10.1038/nclimate1293.

Peng, J., M. Tanguy, E.L. Robinson, E. Pinnington, J. Evans, R. Ellis, E. Cooper, J. Hannaford, E. Blyth and S. Dadson, Estimation and evaluation of high-resolution soil moisture from merged model and Earth observation data in the Great Britain. Remote Sensing of Environment, 2021. 264: p. 112610. DOI: https://doi.org/10.1016/j.rse.2021.112610

Sadeghi, M., E. Babaeian, M. Tuller and S.B. Jones, The optical trapezoid model: A novel approach to remote sensing of soil moisture applied to Sentinel-2 and Landsat-8 observations. Remote Sensing of Environment, 2017. 198: p. 52-68. DOI: https://doi.org/10.1016/j.rse.2017.05.041

Salakpi, E.E., P.D. Hurley, J.M. Muthoka, A. Bowell, S. Oliver and P. Rowhani, A dynamic hierarchical Bayesian approach for forecasting vegetation condition. Nat. Hazards Earth Syst. Sci., 2022. 22(8): p. 2725-2749. DOI: 10.5194/nhess-22-2725-2022.

Shen, R., A. Huang, B. Li and J. Guo, Construction of a drought monitoring model using deep learning based on multi-source remote sensing data. International Journal of Applied Earth Observation and Geoinformation, 2019. 79: p. 48-57. DOI: https://doi.org/10.1016/j.jag.2019.03.006

Singh, R.P., S. Roy and F. Kogan, Vegetation and temperature condition indices from NOAA AVHRR data for drought monitoring over India. International Journal of Remote Sensing, 2003. 24(22): p. 4393-4402. DOI: 10.1080/0143116031000084323.

Smith, R.C.G., J. Adams, D.J. Stephens and P.T. Hick, Forecasting wheat yield in a Mediterranean-type environment from the NOAA satellite. Australian Journal of Agricultural Research, 1995. 46(1): p. 113-125.

Torres, P., M. Rodes-Blanco, A. Viana-Soto, H. Nieto and M. García, The Role of Remote Sensing for the Assessment and Monitoring of Forest Health: A Systematic Evidence Synthesis. Forests, 2021. 12(8): p. 1134.

Tucker, C.J., Red and photographic infrared linear combinations for monitoring vegetation. Remote Sensing of Environment, 1979. 8(2): p. 127-150. DOI: https://doi.org/10.1016/0034-4257(79)90013-0

Upadhyaya, D.B., J. Evans, S. Muddu, S.K. Tomer, A. Al Bitar, S. Yeggina, T. S, R. Morrison, M. Fry, S.N. Tripathi, M. Mujumdar, M. Goswami, N. Ganeshi, M.K. Nema, S.K. Jain, S.S. Angadi and B.S. Yenagi The Indian COSMOS Network (ICON): Validating L-Band Remote Sensing and Modelled Soil Moisture Data Products. Remote Sensing, 2021. 13, DOI: 10.3390/rs13030537.

Unganai, L.S. and F.N. Kogan, Southern Africa's recent droughts from space. Advances in Space Research, 1998. 21(3): p. 507-511. DOI: https://doi.org/10.1016/S0273-1177(97)00888-0

Vanino, S., P. Nino, C. De Michele, S. Falanga Bolognesi, G. D'Urso, C. Di Bene, B. Pennelli, F. Vuolo, R. Farina, G. Pulighe and R. Napoli, Capability of Sentinel-2 data for estimating maximum evapotranspiration and irrigation requirements for tomato crop in Central Italy. Remote Sensing of Environment, 2018. 215: p. 452-470. DOI: https://doi.org/10.1016/j.rse.2018.06.035

Vreugdenhil, M., W. Wagner, B. Bauer-Marschallinger, I. Pfeil, I. Teubner, C. Rüdiger and P. Strauss, Sensitivity of Sentinel-1 Backscatter to Vegetation Dynamics: An Austrian Case Study. Remote Sensing, 2018. 10(9): p. 1396.

West, H., N. Quinn and M. Horswell, Remote sensing for drought monitoring & impact assessment: Progress, past challenges and future opportunities. Remote Sensing of Environment, 2019. 232: p. 111291. DOI: https://doi.org/10.1016/j.rse.2019.111291

Wulder, M.A., C.C. Dymond, J.C. White, D.G. Leckie and A.L. Carroll, Surveying mountain pine beetle damage of forests: A review of remote sensing opportunities. Forest Ecology and Management, 2006. 221(1): p. 27-41. DOI: https://doi.org/10.1016/j.foreco.2005.09.021

Xiao, X., Q. Zhang, S. Saleska, L. Hutyra, P. De Camargo, S. Wofsy, S. Frolking, S. Boles, M. Keller and B. Moore, Satellite-based modeling of gross primary production in a seasonally moist tropical evergreen forest. Remote Sensing of Environment, 2005. 94(1): p. 105-122. DOI: https://doi.org/10.1016/j.rse.2004.08.015

Yao, L., Y. Liu, D. Yang, Z. Cai, J. Wang, C. Lin, N. Lu, D. Lyu, L. Tian, M. Wang, Z. Yin, Y. Zheng and S. Wang, Retrieval of solar-induced chlorophyll fluorescence (SIF) from satellite measurements: comparison of SIF between TanSat and OCO-2. Atmos. Meas. Tech., 2022. 15(7): p. 2125-2137. DOI: 10.5194/amt-15-2125-2022.

Yang, Y., S.S. Saatchi, L. Xu, Y. Yu, S. Choi, N. Phillips, R. Kennedy, M. Keller, Y. Knyazikhin and R.B. Myneni, Post-drought decline of the Amazon carbon sink. Nature Communications, 2018. 9(1): p. 3172. DOI: 10.1038/s41467-018-05668-6.

Zhang, Y., C. Peng, W. Li, X. Fang, T. Zhang, Q. Zhu, H. Chen and P. Zhao, Monitoring and estimating drought-induced impacts on forest structure, growth, function, and ecosystem services using remote-sensing data: recent progress and future challenges. Environmental Reviews. 21(2): 103-115, 2017. DOI: 10.1139/er-2013-0006.

Zhang, F. and G. Zhou, Estimation of vegetation water content using hyperspectral vegetation indices: a comparison of crop water indicators in response to water stress treatments for summer maize. BMC Ecology, 2019. 19(1): p. 18. DOI: 10.1186/s12898-019-0233-0.

K: Drought: A review of the social and behavioural scientific evidence for public messaging

Kevin Collins, The Open University.

Overview

ABOUT THE REVIEW

Effective public messaging around drought in England depends on insights from the social and behavioural sciences to help inform timing, content and delivery of messages.

SCOPE

The scoping review focusses on the published literature and draws on UK and international examples to develop insights into different approaches to drought messaging and their effectiveness in generating a public response appropriate to the message. The review does not include detailed abstract discussion of the different conceptual models of behaviour.

KEY FINDINGS

There is no single understanding of drought nor a single homogenous 'public' for which a unified model of drought and drought related messaging is universally relevant and applicable. Public messaging as a one-way process to alter 'unsustainable' behaviours assumes that what is sustainable is known and rational individuals are primary causes of inefficiency. A more systemic understanding of human behaviour as practice situates individuals within a context and complex interplay of knowledges, experiences, attitudes, technologies, social networks, trust, beliefs, cultural norms and values, and lifestyle. Information is processed through multiple and 'more than rational' routes based on intuition and emotions. Although there is limited quantification of impacts and longitudinal study, posters and information leaflets have minimal impact upon general audiences. Social media, while relevant for short term messages for some, remains one-way and does not reshape discourses and narratives. Messaging from a collective platform of stakeholders is more likely to be trusted than a single author. Irrespective of timing, format or media, messaging which does not recognise or change the practice context will be ineffective beyond the short term. Efficiency framings in current messaging discounts welfare implications and reproduces expectations of the future availability and uses of water.

RECOMMENDATIONS

- Reshape design and content of messaging based on behaviour as practice and multiple ways of knowing and using water in culturally and geographically diverse households and communities.
- Messaging should be flexible and multi-layered in terms of content and characterised by ‘long-term’ actions augmented by ‘short-term’ actions as needed within drought sequence and should convey human agency as a cause of drought.
- Focus on resilience and building adaptive capacity drought as part of systemic managing of the flood-drought dynamic of water as a totality in a climate changing world.
- Rethink messaging as a two-way or multi-actor/node process or system of ‘knowing and acting about drought’ to develop a drought learning system rather than drought messaging system.
- Explore scope for innovations such as citizen science as a platform for two-way drought messaging and learning within communities.
- Research on how reception of drought messaging is shaped by trust in water governing organisations and the potential dissonance of focussing on the individual compared to industry.

1. Aim and structure

To provide advice and recommendations on effective public messaging around drought based on evidence of what has worked and understandings from the social and behavioural sciences.

The report structure is as follows. The scope is set out in Section 2 and methods are summarised in Section 3. Some key terms and framing are noted in Section 4, before the main discussion in Section 5. The conclusions relating to the aim are set out in Section 6 while Sections 7 and 8 provide commentary on current gaps and future research based on the preceding sections.

2. Scope

The work is focused on public messaging about water usage leading up to and during times of drought in England. Included in the scope are:

- timings of messaging during the next winter, spring and summer – both continuity of messaging over time and any seasonal variation
- national scale – all England – and local scales (e.g. within water company areas) messages are in scope. While the geographical focus is England, consideration is also given to other geographical areas that have experienced droughts and messaging for other risks and emergencies as appropriate
- quantitative data on how much water can be saved in the short-term (e.g. the months leading up to and during a drought) to determine the potential size of public water supply savings
- links between messaging and other forms of public communication and engagement should be identified including who is best placed to do what.

3. Methods

The work constitutes a review identifying previous empirical evidence and data on public messaging about drought. This is complemented by research-based insights and theoretical understandings of the issue in the absence of empirical evidence, and also commentary on future research relating to drought messaging.

The work draws upon social and behavioural sciences relevant to various themes including perceptions, motivations, and action. The evidence from the review is used to provide commentary on:

- Key social and behavioural scientific insights about public messaging on drought

- Advice on the timing, content and how messages should be delivered
- A view on the limitations of the evidence base and recommendations for improvement.

4. Context and key terms

Owing to its maritime climate, the UK is generally understood to be a ‘wet’ country. Droughts are intermittent and, while they may be significant, they have tended to be short term. It is perhaps no surprise then that there are more than a hundred words and phrases for describing rain in the English language, but very few additions to the general lexicon associated with drought.

To the extent that language reflects perceptions and concerns, so drought in the UK is a “hidden” pervasive risk, defined and perceived in different ways by diverse stakeholders and sectors – most often as meteorological, agricultural, hydrological, and socio-economic drought (McEwen et al., 2021).

In terms of the general public, concerns about drought have tended to be seen as indicative of a ‘good’ summer, rather than as a developing risk to water supply, human health, food and energy systems and the environment. The dynamics of drought – slow onset and intensification often masked by rainfall – add to the difficulties of communicating about drought to the general public.

Determining ‘effective public messaging’ requires insights into the meaning of effective; who or what is the public; and what is understood by messaging.

For the purposes of this review, the following interpretations are adopted:

- Effective – does the messaging lead to change in behaviours? This does not directly consider efficiency (is the messaging a good use of resources?) or efficacy (does the message achieve its specific purpose such as ‘getting people to turn off the tap’?) but they will be relevant.
- Public – heterogenous in terms of histories, context, social, cultural, gender, residential status, locality, histories and experiences etc.
- Messaging – the wider process or system of communication and learning in the public sphere rather than a singular focus on content elements such as ‘do not use your hosepipe’. Much of the literature reviewed hints at this more expansive conceptualisation of messaging, but tends to be focussed on specific content and encouragement to change water use.
- Drought – an emergent property, phenomena and dynamic process which arises from complex socio-ecological interdependencies. In the literature, the term drought is often reduced to a state or singular ‘thing’, with the implicit assumption of drought as a largely ‘natural’ event or occurrence.

- Drought messaging – in a public context, the provision of advice, information or knowledge to reduce domestic demand and / or change water practices among one or more populations and regions. This may be focussed on voluntary practices and / or pertain to legal stipulations such as introduction of hose-pipe bans under the Flood and Water Management Act 2010.

These interpretations are not assumed to be universal or reflected in the literature, but are important in maintaining a critical perspective on the following.

5. What we know about public messaging

This section sets out what is already known about public messaging in a range of contexts and thematic areas. To identify some social and behavioural scientific insights about public messaging on drought, the first part draws on theories relating to human behaviours, before the focus shifts to more practical aspects relating to strategies.

The following draws on several recent extensive reviews of the literature relating to behaviour change and water or resource related issues. For ease of reading here, the numerous individual citations within any one review to published works are omitted, but can be followed up by reference to the cited review paper itself.

5.1. Defining public messaging

Public messaging is a very broad term. There is no accepted definition other than to refer to content being sent or made available to members of the public.

Depending on context, public messaging could be understood as passing on simple information with a singular focus such as ‘do not use your hosepipe’. It could also be understood as the wider process or system of communication and learning about drought in the public sphere. Much of the literature reviewed hints at this more expansive conceptualisation of messaging, but in practice messaging tends to focus on the specific content and behaviour change elements inviting recipients to alter their use of water.

Public messaging can take many forms from one-off social media tweets and posts, emails, printed information leaflets and signage, to organised events, comprehensive publicity and media campaigns and associated reporting.

5.2. Purpose and assumptions

The purpose of public messaging is often assumed to be to change public (usually household) behaviours in order to achieve or progress some measure of sustainability.

While at one level this seems entirely reasonable, the purpose of public messaging is usually defined from the originator's point of view and may not be 'received' or experienced as such by the public or other recipients. Public messaging may be experienced as something more negative such as 'interference' or 'greenwash' (de Freitas Netto et al., 2020).

The linear assumption that improved public communication or messaging leads to positive public response to water conservation, increases trust in water utilities between a water utility and the public, and help achieve sustainable drought policies is widely questioned (Tortajada and Nambiar, 2019).

When then is the underlying aim of the message? Is it to encourage smaller scale 'everyday' changes to improve 'efficiency' of the current system? An example could be 'only use the washing machine when it is full'. Or is the messaging aiming to fundamentally question existing practices and substantially transform the mental model of how we use water? An example could be 'household use of water is no longer sustainable'. Messaging in either case will be designed, communicated and received very differently.

Within the efficiency framing, messaging tends to be based on the premise that current behaviours lead to an unjustified or unjustifiable waste of resources. It follows that correcting behaviours are needed to promote greater levels of environmental sustainability (Grilli and Curtis, 2021). However, this premise assumes:

1. what is sustainable is known, at least with reference to current behaviours and practices
2. individual behaviours are the primary causes of unsustainability
3. such behaviours exist in all parts of a society or community
4. each individual contributes equally to the unsustainable behaviours and, by extension, to sustainable behaviours
5. those seeking to instigate behaviour change know the right course of action and behaviours to encourage in the public messaging.

Public messaging based on these assumptions risks framing current individual behaviours as 'bad' and new behaviours as 'better' - a framing which carries entailments of judgement, failure and guilt as well as a superior understanding of the situation by others. The framing also risks assignment by the policy and practitioner community of responsibility for the current 'unacceptable' situation to the public / individuals. Such entailments may not be recognised or accepted by recipients of messaging, which could set up an oppositional axis and undermine engagement with the message and messenger. As noted below, expressed ‘unsustainable’ behaviours are not always a ‘free’ or conform to expectations of rational choice. Many other factors may be at play in determining which rationalities and behaviours or practices ‘win out’.

To add to the assumptions, in terms of knowing the right course of action, (Grilli and Curtis, 2021) note that the nature of the targeted behaviour to change is rarely discussed before implementation of the change strategy. As with presuming current behaviours are poor, so it cannot be assumed that the targeted behaviour is the ‘right’ one to target. Not all behaviours have the same impact on environmental quality e.g. reducing waste is better than recycling which is better than throwing away. Which message should be targeted for whom and when? Significantly, the authors note:

> **‘In most published papers, behaviour change applications start from the behaviour to change, not the environmental problem being tackled. Therefore, it is not always clear whether findings represent the optimal solution. When environmental quality is at stake, behaviour change studies should consider the behaviour that causes the largest impact and take appropriate approaches to stimulate behaviour change. In essence focus resources on larger wins rather than the easier behaviours that many have only minimal impact on the ultimate objective.’ (ibid., p.8)**

This insight is particularly important because of the risk of assuming a) the problem situation is known; b) that the intervention being messaged is optimal and will not have unintended consequences; and c) the intervention is meaningful and relevant for the public being targeted. For example, instead of focussing on hosepipe use, public messaging in urban areas with few private gardens may be more effective in focussing on water use in the home itself – i.e. the washing machine and the bathroom.

Behaviour change actions implemented without clear assessment protocols in place will generate limited insight into whether current and suggested actions are successful or not.

Questions about the purpose and underlying assumptions of public messaging give rise to questions about current understanding of behaviours and appropriate messaging.

5.3. Understanding human behaviours

Human behaviour can be defined as the capacity of individuals to respond mentally, physically and socially to internal and external stimuli. Capacity can be potential or expressed. Internal factors (beliefs, values, attitudes, emotions, and knowledge) and external factors (contexts, formal regulation, social and cultural norms) are all important considerations in behaviour change strategies.

There are many models of human behaviour and decision-making across different disciplines such as psychology, education, sociology, anthropology, and economics. These are not reviewed here specifically, but are variously represented in the following discussion. However, there is broad distinction to be made in literature between a focus on behaviours and a focus on practices.

Behaviour change studies have tended to be couched in and focus on the individual framed as economically rational and therefore responsive to rational messaging e.g. ‘please save water because there is a drought’. This includes the Theory of Planned Behaviour (TPB) (Ajzen, 1991) where individuals make behavioural decisions based on rational considerations arising from a strong volition and conscious intention.

As a result, messaging about the environment has tended to rely on a knowledge deficit model – if only people had ‘the facts’ and (often scientific) information then they would behave rationally and more sustainably in accord with the facts. While appealing in terms of simplicity, such campaigns have tended to be monologues rather than dialogues with limited evidence for changes in behaviours (VanDyke and King, 2020). In some cases, individuals are at risk of information overload as the facts are constantly reinforced by high levels of news saturation such as Covid campaigns (Krawczyk et al., 2021).

However appealing, the encompassing singularity of the behavioural intention and the ‘conscious behaviour’ model has limitations and has been widely critiqued in several sectors, including health (Kelly and Barker, 2016), planning (Whitehead, Jones and Pykett, 2011) and energy (Shove, 2014). Essentially, the critique aims to move away from the notion that humans follow a rational model of economic efficiency and maximisation of economic utility, towards a more complex understanding of behaviours where ‘irrationality’ or ‘more than rational’ decision making is accepted as a legitimate position and strategy.

For example, Kahneman (2003) identifies two systems of information processing:

- System 1 - unconscious, energy-efficient, quick, and based on intuition and emotions
- System 2 - conscious, energy-consuming, slow, intentional, and based on cognition.

Rather than conscious and rational processing, System 1 information processing tends to be dominant due to individuals’ lack of mental energy, time, and capacity. At the same time, policy maker’s expectations of public behaviours and thus public messaging has tended to remain within a System 2 framing.

Notwithstanding the different routes individuals use for decision-making and barriers to change, (Kelly and Barker, 2016) in their study of health behaviours identify six basic and persistent errors made by policy-makers concerning their expectations about behaviour change:

1. It is just common sense
2. It is about getting the right message across
3. Knowledge and information drive behaviour
4. People act rationally
5. People act irrationally
6. It is possible to predict accurately.

While people do not always act rationally, it is not true that they always act irrationally. Behaviours cannot be assumed to be irrational within the context of individual lives and experiences. Persistent behaviours are often very functional in that they meet the needs of the individual, however irrational this may appear to outsiders. Hence (Kelly and Barker, 2016) note 'one person's rationality is another's irrationality' and to assume otherwise is the path to arrogance in policy-making and ineffective messaging. Collectively, the six common errors are the stepping stones to a failing communications strategy because the errors embody particular conceptualisations of individuals and behaviour.

In their review of the literature on different Behavioural Influencing Tactics (BITs) targeting long-term water conservation psychology and behaviour within households, (Koop, Van Dorssen and Brouwer, 2019) identify three information processing routes and associated messaging strategies:

- **the reflective route** - the conscious processing of information such as knowledge transfer (KT) and increasing self-efficacy. KT is meaningful and effective only when people understand how they can change their behaviour in ways which they consider feasible. Campaigns which focus on awareness but do not target specific action (i.e. enhance self-efficacy) are less effective in changing behaviour. Increasing the sense of self-efficacy to mitigate droughts leads to actual water savings. Similarly, general information leaflets had no impact on water use, whereas specific labels for appliances resulted in water savings of 23%.
- **the semi-reflective route** – attitudes shaped by rules of thumb and heuristics. Consideration of rational arguments, relevant experiences, and knowledge such as social norms and framing tailored to audience. Public desire in this route is for simple indicators of the best choices which will depend on the credibility of a source, the length of the message, and the behaviour of other people. Social

norms – behavioural patterns that people apply to their own behaviour to conform to their social environment – vary, but research suggests strong social norms coupled with a personal commitment show long term reductions for most groups. For example, shower labels showing ‘x% of your community are abiding by restrictions’ are effective, but can have the obverse effect on already low users who may actually increase their use to match the social norm.

- **automatic route** - choices are made on the basis of an automatic response without the intervention of cognition i.e. ‘done without thinking’.

Associated tactics to encourage water conservation in households for each route are summarised in Table 1.

Table 1 Tactics to encourage water conservation in households for different processing routes. Source: (after Koop, Van Dorssen and Brouwer, 2019:869)

Processing route	Tactic	Principle and effectiveness
Reflective	Knowledge transfer	Providing information to raise awareness, change attitudes, and behaviour. Information campaigns alone often insufficient to achieve long-term water conservation. For temporary water savings, one-sided messages that target high-consuming and relatively uninformed households can be effective.
	Increasing self- efficacy	Enhancing people's belief that they are able to implement the intended behaviour. Providing tips, advice, and concrete examples about how people can save water enhances water conservation behaviour. In particular, short, practical, and timely advice is effective.
Semi-reflective	Social norms	Behavioural patterns that are semi-consciously applied to conform to social environments. Normative messages are effective. Long-term water conservation can be achieved by repeating these messages.
	Framing	Messages emphasising direct impacts, or appealing to intrinsic motivation are persuasive.
	Tailoring	Data-driven personalised messages that increase recipients' responsiveness. Showing how an individual differs from others evokes a feeling of discomfort, triggering water conservation behaviour. Real-time

		information prompts temporary water savings, but there is limited evidence about longer-term conservation habits.
Automatic	Using emotional shortcuts	Evoking emotions in order to influence people's responses. Positive emotions may invoke cooperation and trust, and the use of humour can remove people's resistance. Appeals to fear can also encourage the desired behaviour, provided that people feel high levels of self-efficacy.
	Nudging	The choice architecture that alters people's behaviour in a predictable way without forbidding or limiting freedom of choice. The principle is to make the ‘better’ option more convenient to select. Nudges are rarely applied to stimulate water conservation.

In practice, most BITs are situated along a continuum of all three routes and are dynamic over time. The authors find that after 1–3 months, the high water-saving impacts of BITs begin to fade and water use returns to pre-intervention levels. ‘Early’ BITs used to initiate water conservation behaviour may need to be replaced by ‘long term’ BITs to prolong and reinforce new habits. Combining reflective, semi-reflective, and automatic BITs is likely to be more effective to achieve long-term water saving habits e.g. real-time water use feedback via smart meters generates long-term savings only when such feedback is reinforced by repetition and use of social norms. But social norms or message framing are themselves short-lived (even if applied later) unless supported by tailored feedback or information on imperatives for saving water.

The findings make clear that there is no simple messaging tactic that will work for all because behaviours are complex outcomes rather than simple, rational acts.

5.4. Behaviour as Practice

The move towards understanding the complexity of influences on behaviour has led to an arguably more systemic understanding and approach to behaviour focussed on practice. Arising from social and cultural theory, the idea of practice situates individuals in context amidst a collection of ‘things’ which give rise to certain types of behaviour. Thus,

‘A ‘practice’… is a routinised type of behaviour which consists of several elements, interconnected to one another: forms of bodily activities, forms of mental activities, ‘things’ and their use, a background knowledge in the form of understanding, knowhow, states of emotion and motivational knowledge’ (Reckwitz, 2002:249).

For example, using the washing machine consists of multiple interconnected elements such as: available physical space to accommodate a washing machine; the washing machine and its ‘fixing’ technologies; the user’s understanding of how it works such as programme setting and load capacity; understanding of energy and water use; detergent performance and manufacturer recommendations; household lifestyle, preferences and requirements (e.g. work/school/sports); social and cultural expectations about cleanliness; economic and time elements of convenience and requirement for clothes; availability of piped water; water supply specifications concerning pressure and quality; provision of electricity supply; the price of water and electricity; seasonality and weather; and so on. In other words, using a modern washing machine is a practice rather than a single behaviour of an individual turning it on. In the UK, the practice of using modern washing machines was not possible before the regulation and supply of electricity to domestic units; provision of a piped water supply with standardised fittings; clothing mass manufacturing; mass production of washing machines and so on. Similar elements could be identified for the practice of using a garden hose, leaving a tap running, washing a car and many other domestic uses of water often targeted in public messaging about drought.

Framing behaviour as practice moves away from the ‘common understanding of behaviour as something that is driven by identifiable factors like those of rational self-interest, attitude/motivation or habit’ (Shove, 2014:416). The implications are significant because messaging focussing on making consumption habits ‘visible’ by reference to the need for change does not automatically change the context in which water is used. It also ignores the variety of ‘normal’ consumption practice and different interpretations of the ‘right’ way to live (Chappells, Medd and Shove, 2011).

It follows that trying to change behaviours by simple exhortations to ‘turn off the tap’ or ‘save every drop’ as ‘common-sense’ efficiencies are unlikely to have much effect because they do not fully address the complexity of context and complexity of the elements giving rise to the practices linked to water use.

Expanding this further, Shove’s writing on energy argues that calls for efficiency are counter-productive because they reproduce specific understandings of ‘service’ which may not be sustainable in the long run and also abstracts the resource from the use and production to which it is put (Shove, 2018). In the washing machine example, this could be the expectation that using a washing machine is now ‘indispensable’. Water use also allows transformation of unwearable clothes to clothing useful for work, school or socialising - with all the benefits, rewards and expectations these bring.

Applying Shove’s work on energy to water, messaging based on efficiency risks hiding longer-term trends in demand and use by enabling the status quo to continue. In turn, this reinforces particular configurations of human-environment relations which may no longer be sustainable in a climate changing world. In other words, messaging which aims to maximise water efficiency can mask deeper, unsustainable patterns of water use. By way of example, in the UK the daily average direct use of approximately 150 litres of water per person has been growing every year by 1% since 1930 (Water UK, undated) despite many water saving efficiencies. On this basis, the practice of water use in households might be considered unsustainable and no amount of incremental efficiency gains through messaging will recover. Indeed, it could be argued that messaging which continues to 'allow' the current framing of water use is missing the point.

As part of growing acceptance of the Anthropocene (Savelli et al., 2022) reviewed recent drought literature and how social processes are conceptualized and incorporated in physical and engineering studies. They found that research increasingly 'recognizes society as an active force rather than just a passive subject during the occurrence of drought events'. But they note researchers continue to conceptualise society as homogeneous entity that, as economies and populations expand, will continue to consume more water. The authors note that this framing of society is misleading because society is heterogeneous and growth is unequal. Moreover, framing decision-making processes as apolitical 'conceals the power relations underlying the production of drought and the uneven distribution of its impacts' (ibid.). In practice, only some social groups contribute to drought hazard and propagation from a meteorological into a socioeconomic drought. Existing studies rarely consider these dynamics or understand that alterations of water flows are not the causes but rather the symptoms of anthropogenic changes.

In terms of messaging, (Savelli et al., 2022) findings are important because they emphasise the heterogeneity of society; the anthropogenic dimension of drought; and need for critical awareness of how power relations shape uneven drought propagation and impacts. In other words, public messaging needs to recognise that drought cannot be framed as simply a hydrological or climatic event and that different social groups will be affected differently by drought impacts and related possible water restrictions.

Following (Badullovich, Grant and Colvin, 2020)'s exploration of reframing climate change, reframing drought through public messaging offers an opportunity to reframe behaviour change. According to (Lamm et al., 2018), behaviour change requires understanding how the problem of water is framed as a precursor to understanding motivation and in turn designing communication strategies. Using Situational Theory of Problem Solving (STOPS) theory to understand water irrigator behaviours in the US, they identify three independent variables to explain and predict behaviours in a particular situation:

1. problem recognition - an individual's cognitive awareness of a discrepancy between expectations and observed 'reality'. Lack of awareness of a problem and related information means there is de facto no problem or issue to address.

2. involvement recognition - an individual's perception that he/she is connected with a problem;

3. constraint recognition – an individual's perception that obstacles exist that limit their ability to act.

The three variables contribute to the degree of motivation to (i) find out more and (ii) the transmission [sic] of information and understanding to others in the specific or similar situations. High levels of problem recognition but low perceived levels of involvement lead to a disconnect or cognitive dissonance between irrigators' behaviours and their connection to water conservation (Lamm et al., 2018).

The research also found a negative correlation between sense of involvement and 'talking with others' about their practices. In accord with other studies, they speculate that irrigators are aware of their 'unenvironmental' practices, but still continue out of economic needs or lack of alternatives and are therefore uncomfortable discussing it and communicating with others.

The researchers suggest designing communication strategies based on a community-scale [compared to an individual] model to show how individual actions shape and are shaped by the situation as a way of avoiding specific blame while also encouraging problem and involvement recognition.

The extent to which this also applies to households / public in terms of messaging is uncertain but offers scope for experimentation at locality and neighbourhood level. It also points to the wider question of rethinking how drought is framed in messaging and widening the scope for messaging conventionally focussed on individual actions.

5.5. Barriers to behaviour change

While understanding behaviours as practice offers many insights, there remain many barriers or constraints to change in relation to drought (Blair and Buytaert, 2016). From their review of the literature, (Grilli and Curtis, 2021) identify the following key barriers:

- Attitudes and temperament (e.g. laziness) especially in people with weak environmental concerns
- Responsibility - individuals do not engage in environmental behaviours because of a lack of trust, leading to a belief that their individual behaviours cannot influence the situation
- Practicality - social and institutional impediments e.g. lack of time, money and information impact on behaviour regardless of individual attitudes.

Making sense of the barriers facing people is a fundamental step in behaviour change implementation when designing approaches and interventions and public messaging.

Accordingly, (Grilli and Curtis, 2021) suggest any behaviour change programmes should consist of the following:

1. Select the behaviour to change. Behaviour change aims to achieve the largest positive impact on environmental quality, but not all behaviours are equally important. Small changes in key behaviours may be more beneficial than very large changes in secondary behaviours. Behaviours should also be clearly defined, to avoid misinterpretation of actions needed.

2. Choose approaches based on type of behaviour objectives. All interventions are potentially successful if implemented in the right situation. Intervention success is context-dependent and the most appropriate should be selected on the basis of the chosen behaviour and objectives.

3. Establish a control group. Comparing groups that only differ by the intervention may be helpful to precisely identify intervention effects and behaviour change success.

4. Start small - In most cases it is difficult to establish a priori the most suitable approach to behaviour change. Small pilot studies may be appropriate.

5. Define the measure of success and measurement methods.

6. Define the outcome to measure. Inputs should not be confused with outcomes. Measuring changing behaviours may be insufficient to achieve the desired environmental outcome, particularly if the targeted behaviour is poorly selected. Measurement of change in behaviours may offer an early insight on whether the ultimate goals of the strategy might be achieved in the future. However, if behaviour change is negligible, subsequent causal improvement in the environment itself should not be expected.

7. Record resource inputs to quantify relative efficiency of approaches and adaptation of approaches.

8. Monitor the long run. Collect performance data regularly to assess policy impacts of the measures and provide new stimuli as needed. Follow up approaches may vary.

Of course, this listing is an ideal and, in practice, some requirements such as long run monitoring will be particularly difficult to achieve. However, the authors stress that behaviour change is just one of the several concurring causes to environmental degradation. They make clear it can be misleading to overly emphasise the role of behaviour. In some situations, socio-institutional barriers are dominant and behaviour changes are less effective. For example, energy-efficient behaviours would be not so important if all energy was produced from renewable sources. Correspondingly, in the water sector, water quality campaigns for rivers and coasts would not be required if water companies and water organisations were addressing combined sewage overflows. When individuals have no control over several of the factors affecting environmental problems, governments can be more influential because they possess public policy tools to effectively tackle environmental problems and redirect human actions towards more sustainable paths.

Similar findings emerge from a review of the literature on the social dimensions of urban water consumption during droughts in developing and developed countries (Asprilla Echeverría, 2020). Gaps between intention and effective behaviour in conserving water in households is linked to the social dimensions of urban water saving patterns arising from:

1. Price inelasticity of water leading to price increases disapproved by users unable to reduce use
2. Increasing domestic consumption despite water saving devices
3. Differences between a household's understanding of real water consumption; the reported household consumption level; and desired consumption each household wants others know
4. Gaps between desired and actual practices
5. Waning of water reduction practices over time

Accurate predictions of how individual people will behave in any given situation therefore remain elusive. In a health context, (Kelly and Barker, 2016) note that identifying larger, population-level changes and patterns of behaviours is possible, but they suggest this does not lead to tools or understanding of how to tackle health in-equalities, the obesity epidemic or increases in alcohol consumption. A key barrier is that policy-makers do not engage with the complexity of behaviour at individual level and risk relying on platitudes about 'getting people to change their behaviour'.

5.6. Behavioural bias

The complexity of behaviour or practices arises from many factors including bias. Although definitions vary, bias can be broadly understood as behaviours which do not conform to expected or logical patterns. Understanding behaviour bias can provide insights into how and the extent to which drought messaging is understood and acted upon to inform possible future messaging.

In their comparison of COVID-19 and climate change and lifestyle responses (Botzen, Duijndam and van Beukering, 2021) identify similarities in causes and consequences of both phenomena such as greater impacts on deprived and vulnerable communities.

Based on research from psychology and behavioural economics, they suggest the pandemic and climate change can be characterized as low-probability–high consequence (LP-HC) risks, where individual behaviour deviates from rational risk assessments by experts and optimal preparedness strategies. Risk-related behaviour biases in the context of individual decision making include:

1. **Simplification** Individuals are likely to make choices by focusing on either the low probability of a disaster occurring or its potential consequences, instead of making a 'rational' assessment of the full risk distribution. Many people use threshold models to decide whether to take protective measures in advance of a potential catastrophe. Climate change-related risks, such as natural disasters, have a low probability that individuals simplify to being zero or falling below their threshold level of concern. However, when the event occurs in their surroundings individuals will start to focus on taking adaptive measures. In drought context, the future risk of drought may be reduced to zero and ignored until the drought is actually impacting locally and directly.

2. **Availability** Individuals underestimate LP-HC risks until after they or friends or family experience the consequences of the event. Until that point, they are reliant on readily (and often recently) available information as fact. Availability bias is especially problematic in the case of environmental risks, which are not generally salient until it is 'too late' to reverse trends – a common problem with drought.

3. **Finite pool of worry** is when increasing concern about one issue causes a decrease in concerns about other issues because individuals only have a limited pool of emotional resources. Concerns about climate change tend to 'fall away' during financial crises or national emergencies such as Covid. As with availability bias, drought may only be ranked highly in a ''finite pool of worry" when it is too late to prevent severe impacts. It will also fall away rapidly when rainfall returns and flood risks increase.

4. **Myopia** occurs when individuals (and organisations) evaluate investment decisions over shorter time horizons than required, before investments in climate change adaptation and mitigation yield positive returns. Reductions in future risk and benefits are discounted in favour of upfront costs, thereby reducing investment in long term climate change mitigation and adaptation measures. Short-sightedness ensues.

5. ''**Not in my term of office**" refers to the political reluctance to undertake expensive measures to limit low-probability risks because there is little reward from voters for limiting the impacts of events that do not occur when they are in office. Also voters tend to reward politicians who 'save' them from an extent emergency, rather than avoid it in the first place. This may be particularly significant for public and political support for long return droughts where

immediate costs may be high, returns are low and benefits may accrue many years later.

6. **Herding** bias occurs when individuals' choices are shaped by others' behaviour, especially under conditions of uncertainty due to social norms. For climate change related risks, individuals are more likely to take measures to limit the damage from natural disasters when friends or neighbours have or are adopting similar measures. Triggering or emphasising social norms, for example, by highlighting the climate-friendly behaviour of others, especially friends or neighbours, may be an effective means of stimulating climate action.

For each bias, (Botzen, Duijndam and van Beukering, 2021) make the following recommendations for communication policies:

- Simplification and availability biases - communication strategies should stress the consequences of risks to individuals. Cognitive dissonance can be limited by, for example, using constructive framings and personalizing climate issues so they are perceived as less distant. For example, Bradt (2019) showed that demand for flood protection increased after people received information on the consequences if they were to personally experience flooding caused by hurricanes. However, drought may be more difficult to identify personal consequences as water supplies will be prioritized and hose pipe bans may be introduced regardless – i.e. there is no clear choice to be made about the consequences of watering gardens. For an urban community with small or no gardens, the personal consequences of drought will tend to be even more minimal until it begins to affect food stocks and prices and / or an amenity. In which case messaging to address other biases may be more appropriate. Communication strategies that emphasize human health risks, in particular, may be effective in enhancing ongoing support for drought and flood risks over the long term, and especially where personal consequences of drought are less evident in urban areas.
- 'Myopia and NIMTOF biases' can be overcome by linking and emphasizing interconnections between policies and measures that are currently adopted to limit the risks of drought to actions that also reduce the risks from climate change or human health or food security.
- Finite pool of worry - drought messaging should focus on acting more sustainably in environmental and economic terms, with corresponding benefits to humans.
- Herding bias - could spread positive drought behaviour – though messaging would have to emphasise who is already taking action among a community the target audience would identify with and trust. This could be targeted at neighbours and friends and existing social networks.

While these suggestions have merit, they are not easy to implement for a range of reasons, including the difficulty of differentiating between different recipients and user groups.

5.7. Differentiation of public

The term 'public messaging' carries a sense of homogeneity of recipient. As is widely recognised, in practice, the public is highly diverse. Communication and marketing campaigns typically identify, or segment, audiences based on a range of demographic indicators such as socio-economic status, purchasing preferences and ethnicity – the exact mix depending on the aims and context of the campaign.

After the UK 2018 drought water companies have continued to rely on traditional demographics, although there are some nascent efforts to understand their customers in terms of attitudes and values such pre-disposition to the environment and conservation (Larbey and Weitkamp, 2020).

Based on research work in Australia, (Dean et al., 2016) explored differentiation within a population on engagement in water-related issues. Using criteria of demographic and household, life experiences and support for policy initiatives, they identified five key groups and typical characteristics:

1. Disengaged - typically younger, male, lower socioeconomic status, rural
2. Aware but inactive – younger male, lower income, renting, no garden, urban
3. Active but not engaged – female, some qualifications, renting, experience of water restrictions, garden
4. Engaged but cautious – older, higher income, home-owner, longer residency, garden, participates in community, rural
5. Highly engaged – female, higher income, tertiary educated/trade, home-owner, English speaking, changes behaviour, aware of information campaigns about water.

Characteristics relating to homeownership, gender, ethnicity, education, having a garden, being older, and experience of water restrictions all had a significant impact on each of the water policy engagement profiles.

As might be expected, the authors also found that sensitivity to social norms (others save water so I should do the same) was also higher in the more engaged groups, extending to awareness of injunctive norms (others want me to save water) in the highly engaged. Ethnicity may shape water practices not just because of language barriers, but because of cultural traditions of understanding and using water. Earlier research in the US identified similar sets of factors and recommendations (Adams et al., 2013) while (Tijs et al., 2011) found distinct differences in support for water conservation between rural and urban contexts. Rural areas experiencing drought readily adopted behaviours compared to urban areas.

(Dean et al., 2016) suggest the following strategies for interventions with different groups.

- ‘Disengaged’ individuals - will tend to be focussed on everyday routines which may include managing the impacts of low socio-economic status. Without gardens and renting, the main emphasis for this group will be to develop relevant information for young, male, urban renters with families during times of restriction to help develop a foundation for new water practices. Building social capital and community support is likely to be important perhaps through community gardens or similar.
- ‘Aware but Inactive’ - young, male, urban renters already have some water-related knowledge but limited behaviours. Engagement initiatives for this more educated group should focus on everyday indoor behaviours rather than garden-related as a means to developed longer term identity and social norms related to water use.
- ‘Active but not engaged’ – gardening can be an important motivator for water conservation to continue gardening. Financial savings may also be relevant. Messaging reinforcing the effectiveness of everyday behaviours can help develop a sense of environmental identity and social norms.
- ‘Already engaged’ - messaging to promote innovative behaviours and supporting with policy; reinforcing existing behaviours; enhancing the sense of environmental identity; encouraging active social networks, and strong reinforcement of social norms.

The suggestions are directed towards the Australian context, but the findings are broadly relevant to the UK, especially the need for messaging which recognises different levels of engagement and differentiation of education, income, age, ethnicity, dwelling type and location i.e. urban / rural.

Even within an urban context, the particulars of the urban water supply systems are also significant. During Australian droughts, the sense of crisis was greater in some cities than others and led to different household water conservation practices. Compared to Sydney and Melbourne, the city of Perth - with access to alternative water sources through desalination plants and household boreholes – displayed weaker household water conservation behaviours (Lindsay, Dean and Supski, 2017).

Similar variations exist in commercial contexts where adaptations of water technologies do not guarantee water reduction. For example, Canadian farmers adapting to drought through behaviour and technological changes went on to increase the area they irrigated, leading to a rebound effect (the so-called Jevons paradox) on water consumption (Ghoreishi, Razavi and Elshorbagy, 2021). As with the pervasive notion of renewable electricity being ‘free’ and ‘unlimited’, in the UK proposals and strategies for increased desalination and increased inter-basin transfers may actually undermine longer-term water conservation behaviours.

The implication of the above is, for example, that urban, culturally diverse populations of widely varying socio-economic status are likely to require substantially different messaging tactics and approaches compared to more rural populations. Messaging needs to reflect the characteristics of households, be relevant to locality (city/urban/rural) and sensitive to cultural differences. This could also be linked to ingroup norms (see below).

5.8. Messaging tactics and strategies

In their review of 85 case studies from the literature on pro-environmental behaviour change, (Grilli and Curtis, 2021) categorise 5 main behaviour change methods and strategies:

1. education and awareness
2. outreach and relationship building
3. social influence
4. nudges and behavioural insights
5. incentives.

These headings provide a convenient basic structure to explore messaging strategies from a range of authors with the caveat that a spectrum of strategies is likely to be more effective. However, almost without exception, the literature does not quantify specific water savings (or otherwise) from messaging interventions.

5.8.1. Education and awareness

Education and awareness raising activities vary from passive, e.g. posters, leaflets, to more engaging activities, e.g. public events and activities such as ‘conservation work groups’. The more passive forms tend to be popular because of the relative ease and limited resources necessary. However, as a strategy it tends to have it has the lowest success rate and limited impact on energy and water conversation behaviours (Grilli and Curtis, 2021). In general, top-down programmes telling people “what to do” are ineffective, although (Katz et al., 2016) found that information based water conservation campaigns in Israel were more effective than price signals, especially where augmented water supply (e.g. desalination) is significant. Although a very different context, the experience of Israel’s water system is relevant to the UK if increased desalination and also augmentation in the form of a national water grid is enacted.

Tailored rather than generic information is specific to the user and context relevant. As an example moving towards more tailored information, Affinity Water’s information and education campaign during 2022 and 2023 has focussed on information relating to specific catchments and rivers geographically relevant to users who are encouraged to save water to protect 'their' river and water supply. There is no data to determine if this has been successful. However, as noted below, where performance is contrasted with social norms or other groups, is can be effective for encouraging behavioural change, although selection of the comparators is critical (Grilli and Curtis, 2021).

In terms of themes to emphasise, 'more efficient use of resources' offers some scope, but this is not widely appealing to most user communities. As noted above, efficiency is largely a technical framing and lacks resonance with the complex practices of households who may prefer social and cultural outcomes associated with water use rather than water efficiency per se. One case study of farmers and their water practices found the combination of tailored information and the making of public commitments was especially effective in eliciting behaviour change, even in a one-off meeting (Grilli and Curtis, 2021). However, public commitment is hard to engender and replicate in most passive public and household education and awareness strategies.

Even with information, however, a lack of resources and infrastructure relating to the desired behavioural change can result in a sense of helplessness by recipients. Unable to make their own decisions about using water in domestic context (i.e. feeling of having no real choices) leads to disempowerment and disengagement.

5.8.2. Social norms

'Social norms' constitute individuals' beliefs about what a majority of other people do, or approve of doing. An individual's sense of social identity is shaped by the process of psychologically belonging to a group – known as the ingroup. An individual's own attitudes and behaviours tend to reflect the norms advocated by the ingroup.

Research in the US on water reduction by households found that provision of water saving tips and general appeals to prosocial behaviour had short-term impacts on water use patterns i.e. less than 12 months (Ferraro, Miranda and Price, 2011). Messages which combined prosocial behaviours with social comparisons had a much longer impact on water demand (extending beyond 24 months). The research incorporated strong social comparisons of norms on water bills such as the following:

'As we enter the summer months, we thought that you might be interested in the following information about your water consumption last year: Your own total consumption June to October 2006: 52,000 gallons. Your neighbors' average (median) consumption June to October 2006: 35,000 gallons. You consumed more water than 73 percent of your Cobb County neighbors.' (Ferraro, Miranda and Price, 2011:319).

While the authors provide good evidence for the use of social comparisons, they note future research should explore the short-run and long-run welfare implications of using norm-based strategies. Just as with social impacts of current energy prices, there can be many reasons why some households are unable to reduce water use; and some households may feel under pressure to reduce further at the expense of their quality of life.

The findings are reinforced by research on social norms for water behaviours in SE England which suggests that messaging based on an appeal to people's social motivations and providing information regarding the activities of other people may be more effective (Lede, Meleady and Seger, 2019). This is especially so if the communication focusses specifically on an 'ingroup' rather than a more general 'others'. As with the US example above, the norms of relevant ingroups were expressed as comparison such as 'other guests in this hotel' or 'your colleagues in X organisation' or 'other households in your locality…' undertook Z behaviours to reduce water use. An example from their study is shown in Figure 1.

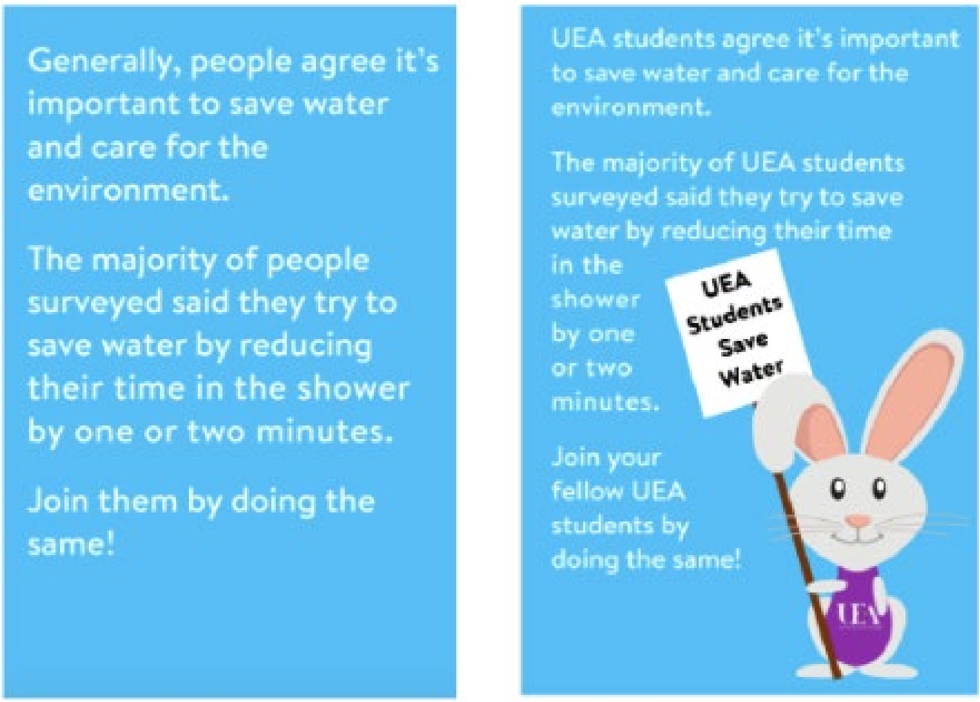


Figure 1 Waterproof shower stickers used for general social norms (left) and the ingroup norms (right). The ingroup norm was more successful (Lede, Meleady and Seger, 2019).

However, the emphasis on social norms is not without problems, in particular the 'boomerang effect' (Hart and Nisbet, 2011) where the intervention or message creates the opposite action leading to the 'total collapse of sustainable behaviour' (Berger, 2021). For example, (Lede, Meleady and Seger, 2019) report that pro-environmental behaviours are weaker when individuals compared their own national group (UK) to an outgroup with a perceived superior environmental record (Sweden), and strengthened when comparing to an outgroup perceived to be less environmentally conscious (USA). In either case, existing low users also tend to increase their consumption to the general social norm.

5.8.3. Thresholds

Thresholds are a key part of drought modelling, but their use and effectiveness in public messaging is still uncertain. Recent research explored use of thresholds in farmers' perceptions of drought in Scotland. The findings suggest

> **'thresholds for public/community awareness and action will vary subjectively depending on a variety of factors, including the nature and extent of people's connections to signs of emerging drought, with the "most severe drought" determined by their activities and goals at the time of the event' (McEwen et al., 2021).**

The research suggests combining scientific thresholds with local threshold perceptions arising from the lived experience of an environment and drought might help to ensure the nature of the message is more reflective of localities and local interests. This could assist with adaptive and resilient behaviours. Citizen science could be used as part of understanding droughts and incorporating this within messaging and thresholds as a future research avenue. The implications are that messaging incorporating thresholds is a promising area, but will need to reflect the complexity of and sensitivity to different perceptual thresholds and levels of risk aversion.

5.8.4. Social influence

Social influence is one individual effecting a change in the behaviour of another and often arises in close social groups such as family, friends and neighbours (Grilli and Curtis 2021). Interventions via social influence are flexible and include:

- leaders - often volunteers within the social network of targeted individuals who convey information about the issues.
- public commitments e.g. to reduce water use in the garden are particularly effective when made in public.
- model behaviours – an 'model' individual engaged in a certain behaviour which other individuals emulate
- feedback – individuals or groups receive feedback on other people's behaviours.

(Grilli and Curtis 2021) report the literature suggests that the face-to-face element in social influencing methods (leaders, public commitments, model behaviours) tend to be more effective than methods with just feedback, and tend to be more successful than education-based initiatives in stimulating specific pro-environmental behaviours. However, assessment of the long-term durability of social influence studies is poor, and there are indications that the effect is lost within 12 months. More research is necessary for conclusive findings and repeated interventions may be necessary.

The use of social influencers is a more recent phenomenon on social media though celebrity endorsement and publicity has often been a feature of more conventional conservation marketing campaigns. Evidence of the impact of celebrity influencers is limited, but a study by (Duthie et al., 2017) of UK-based conservation organisations using celebrities found that celebrity endorsement can result in positive and negative effects. Although there is an increased interest and engagement with an advert featuring a celebrity, this varied according to the characteristics of the celebrity and the respondent. Recall of messaging content also varied considerably and was actually lower with well-known celebrities. There is no single celebrity that appeals to all sectors of the 'public' and the celebrity can overshadow the message itself. More research is needed on celebrity endorsement and the role of social influencers and how to evaluate their impacts.

5.8.5. Outreach and relationship building (ORB)

ORB interventions such as focus groups, training sessions and public meetings are most successful principally because they combine the opportunity for positive relationship building as well as education and social influence. They are more likely to be effective where the sense of belonging to the community is strong.

However, investment in time, resources and community liaison is significant and because ORBs are performance-oriented, often responding to specific needs, the scientific aspects are often considered less important. Measuring success varies from numbers of participants, numbers of time or changes in behaviours such as changes in recycling rate or energy use, however this is also difficult to measure unless the ORB involves some ongoing commitment or follow-up surveys. The research suggests that ORB events with very defined purpose have stronger impacts on behaviours, i.e. a workshop on 'how to save water in your home' is more likely to have a stronger impact than a 'Learn about the environment day'.

5.8.6. Nudges

Nudge interventions can be successful, especially where small changes to the 'choice architecture', e.g. number, location and size of recycling bins for waste, has a positive impact on pro-environmental behaviours. However, nudges can be resource intensive and behaviour changes are not likely to be permanent when the nudge is no longer in place.

In one of the few documented studies, a 10-year case study of Singapore used anonymised billing data from 1.5 million accounts to explore the effects of a national Home Improvement Programme to increase efficiency of residential water plumbing compared to nudging. The plumbing improvements led to a 3.5% reduction in water consumption which persisted for over a decade. The comparative nationwide nudging programme found no evidence of reduced consumption, leading to the conclusion that 'plumbing can be changed, but people cannot' (Agarwal et al., 2022). The research suggests that thinking about behaviours in terms of practices offers scope to undertake more effective interventions than just messaging.

The efficacy of nudging aside, the ethics of nudging have also been questioned (Shove, 2014), (Kuyer and Gordijn, 2023) because they are not grounded in the traditional economic welfare theory or subject to political scrutiny and accountability in the same way as an extant policy. It is for this reason that nudges can be considered manipulative, although complete transparency about the nudge can help overcome some of these concerns.

5.8.7. Incentives

Incentives are widely used in the waste, energy and water sectors. The effectiveness of monetary compared to non-monetary measures is still subject to debate. In some cases, non-monetary incentives yielded a larger impact, but in other cases the reverse. While offering concrete rewards, incentives do not stimulate individuals' awareness towards environmental problems and in some cases may decrease intrinsic motivation as an attitude of ‘I will only change if I get paid’ becomes more entrenched. It is also likely that behaviours are not maintained when incentives are removed. Incentives may be particularly appealing for local authorities, because pro-environmental behaviours are related to provision of public services and budget constraints, e.g. recycling.

From their analysis of the impacts of the different approaches, success rate and area of application, (Grilli and Curtis, 2021) suggest that no single approach is preferable by default. Success depends on the organisation of the intervention and the choice of approach should be determined by specific objectives and available resources.

In broad terms, outreach and relationship building (ORBs) appears particularly effective for topics of general interest, such as climate change or sustainable lifestyles, and usually are not targeted at specific behaviours, where education or social influence approaches may be more appropriate. There is no conclusive evidence that incentives outperform other behaviour change approaches and more research is necessary for their wider support in behaviour change activities. Nudges for individual behavioural changes can be effective in the short term, although ethics may be a concern and their use is limited to instances where the choice architecture is modifiable. The Singapore example suggests more ‘hardwired’ water reduction interventions deliver longer term results.

All of the messaging strategies and tactics are suitable but specific objectives and the target population will influence which should be used. For example, people who consider themselves sustainability-oriented is a determinant for some, but not all pro-environmental behaviours. Behaviour-specific self-identity (e.g. pro-water sustainability) has greater force in behaviour change than the more generic ‘pro-environmental’ self-identity. Additionally, past behaviour independently exerts a strong influence on behavioural intention.

The findings suggest a multi-stage approach using different messaging strategies and interventions over time may be more appropriate, but evidence of the effectiveness of messaging in any of these strategies over the longer term is questionable and requires further research.

Encompassing many of the above insights and recommendations, the DEFRA-commissioned RADAR project reviewed existing approaches to drought risk communication (DRC) in the UK, Europe and more widely. While the study focussed on DRC rather than public messaging per se, the findings are relevant.

As already noted, the language of drought is problematic, and definitions of drought risks vary by sector, context and geography. Public understanding of drought and drought processes – at least from a scientific and technical point of view - remains fragmentary.

Approaches to communicating drought are very diverse, and while narratives and storytelling can be effective at fostering behaviour changes, statistics are also embraced by professional audiences. Different levels of detail are needed according to audience interest. Depending on timing, social media, emails and news reports were more likely to reach and impact audiences rather than lengthy drought reports, although even the impact of these more immediate messaging strategies may be limited.

The RADAR project suggested key actions for public messaging about drought include the following:

- Ensure localised communication in order to be relevant
- Shape and contextualise the range of risks and impacts
- Communicate uncertainties and likelihoods, as well as impacts that are understandable
- Provide meaningful, relevant and timely data to different communities
- Use appropriate language and terminology – the word “drought” does not resonate with all members of the UK public, but other terms such as “prolonged dry weather” can be seen as ambiguous
- Review the tone, empowerment and ambition of the communications – is it aiming to and how should it support action?
- Encourage individual and community response to drought risks
- Complement and enhance existing drought planning arrangements and, by extension, existing water governance strategies
- Establish clarity on timing of messages.

(Smith et al., 2021)

The overriding message from the RADAR review is that the complexities of drought definitions, language, measurement and phenomena in highly contextualised situations with segmented audiences and potential collaborators presents significant challenges to public messaging about drought risk. Importantly, the RADAR findings suggest drought and the risk of drought needs to be reframed from a “thing” or a singularity that is communicated, towards being at least a process or a “system” that has many inter-related elements.

5.9. Media and format

The media, format and design of messaging refers to the various aspects associated with the delivery and structuring of the message rather than the specific content, although there is clearly a relationship between the two since some media formats limit the content which can be messaged.

Public messaging in a media rich context such as the UK means it is not possible to comment on all of the possible permutations available for public messaging. The ethical issues concerning digital divides - often linked to age profiles of the intended audiences – also vary according to media and format. Notwithstanding, in general terms, evidence on the successes of different media and format used for messaging is patchy and largely unquantified.

5.9.1. Posters, leaflets, articles

Messaging in this format can be extremely varied, ranging from mostly text-based, an infographic or mostly visual materials. Messaging can be a one-off or as part of a campaign such as additional information and regular updates added to users’ bills. While useful as reminders or sensitisers to an issue, particularly for adults with less online connectivity, as noted above, the effectiveness of posters and information leaflets is generally considered to be low (Grilli and Curtis, 2021).

5.9.2. Emails

Email based communications and campaigns are also increasingly used by some water companies for regular updates and messaging on floods and drought risk directly to their customers. There are few studies of their effectiveness, although in the health sector (Agachi et al., 2023) report regular emails originating from health apps were cost effective, but had limited impact on changing behaviours relating to physical activity. The literature is generally silent on the effectiveness of email-based campaigns. This may be because the focus about messaging has shifted to the use and role of social media.

5.9.3. Social Media

The role of social media as a more targeted and accessible platform(s) for messaging has generated considerable interest, but with mixed findings.

A study of drought communication in California's 2014 drought noted that social media served multiple purposes: one-way information sharing, two-way information sharing, situational awareness, rumour control, reconnection, and decision making (Tang et al., 2015). Of the various social media strategies, the most effective communication to deliver largely one way drought risk information to professionals and the public was focussed on a particular theme – in this case: 'Save Our Water'. Significantly, the social media channel used for this was supported by regulators and water agencies. Posts were personalised and carried specific messages alongside visual materials.

The same study found that Twitter was effective in disseminating drought information into existing social networks. They also find that Facebook has enabled two-way information sharing for drought risk information and water conservation strategies, but the authors concede that more research is needed to understand the use of social media, user contexts and data quality. More research is also needed on the extent to which social media users' input and responses were considered and incorporated into subsequent decisions on drought risk management. In other words, there is limited evidence for social media being about 2-way communication rather than just the more passive 2-way sharing.

A review of the use of Facebook by 20 large Australian and U.K. water businesses between 2010-2017 noted that using social media helped to promote the health benefits of tap water, water conservation behaviours and responsible wastewater practices (Pawsey, Nayeem and Huang, 2018). However, the research found that most firms made less than one post per day. Similarly, of the customers who engaged, most did so with a single response which suggests limited impact. The study concluded that more research is needed both on the effectiveness, age-group profiles and the degree of digital divide associated with messaging via Facebook and other social media.

A more recent study of public communication on water availability and water conservation actions in the Republic of Ireland (ROI) focused on the use of social media (Twitter and Facebook) and newspaper publications in drought years of 2018 and 2020 (Antwi et al., 2022). The authors note that in the ROI, drought communication has become 'laborious' principally because of previous communication approaches, public perception of water resources availability, differences in understanding how water supply is shaped by over-abstraction, climate change, mean precipitation and evaporation; and media coverage of drought or climate change events. In other words, social media messaging is not done in a vacuum, and no matter how slick the media and format, the history, context and content of messaging to date are key. Social media is no guarantee of success in focussing attention or behaviour changes in isolation.

For example, while some communications by the national utility were considered positive, its engagement on social media was considerably limited. Newspaper coverage of drought and water availability was greater in 2018 compared to 2020, but in both print and online messaging, the prevalent framing was uncertainty and risk.

Although the lack of a national drought plan or drought information system in ROI is somewhat different to England and Wales, the authors suggest that for public messaging:

- drought resilience is best framed as a proactive approach rather than a crisis response
- public engagement strategies and collaborative efforts to communicate drought and water conservation should be led by coalition or forum rather than individual water companies. This would also involve active stakeholder engagement involving state institutions and the private sector, individuals, academic and financial institutions before, during and after drought to encourage commitment to long-term actions and resilience building.

In a comparative analysis of the 2017-2018 Cape Town Drought and 2019-2020 UK Winter Floods (Mcalpine, Sahar and Pannocchia, 2021) assessed the strategies for social media public messaging to promote social behaviour change by at-risk communities. They found high levels of convergence in terms of content and messaging between the UK and Cape Town, but differences in channels.

In both Cape Town and the UK, public and water authorities published messages from their own organisational accounts. In the UK, there was also marked cooperation between multi-level stakeholders in social media campaigns to run parallel campaigns focusing on aspects relevant to their operations. A shared Facebook page was also set up to disseminate flood resilience content. In the Cape Town drought, posts from private companies, community organisations, water suppliers and public authorities were posted natively, i.e. on their own media channels rather than shared but also shared as part of a unified (cf to the UK parallel) campaign to persuade the public to abide by restrictive water rations.

The other main difference between the SA and UK events were:

Both online campaigns used multi-media to convey their messages, including video, images, infographics, memes and employed common hashtags. Tips and advice for saving water and improving flood resilience were frequent and posts also included hyperlinks to re-direct audiences to digital toolkits and resources. Additionally, both cases launched social media competitions and challenges to improve engagement with content, uptake in the campaign and spread messages through social networks. Common messaging strategies included public sensitisation of risks posed by droughts (Day Zero) or the dangers of driving through flood waters.

Aiming to change public patterns of behaviour, the posts focused on enhancing self-efficacy: how to reduce individual water usage in Cape Town; and adopting personal protective measures and avoiding risky behaviour in the UK, such as driving in deep water. Solidarity messaging in Cape Town included a collective objective to avoid ‘Day

Zero' through individual actions, while in the UK the emphasis was on community readiness as well as individual actions. In both cases, messaging used culturally relevant norms and values of the respective target audiences.

The findings suggest a shared platform or account for unified drought messaging alongside individual or native organisational messaging may be important for developing trust and visibility (the Disaster Emergency Committee of stakeholders is perhaps an example of a visible and trusted platform). A focus on enhancing self-efficacy and reliance is important, although it is notable that both examples are of very tangible risks. Flooding in the UK or the Cape Town emergency relating to imminent loss of public water supply is very different to the more usual diffuse nature of drought in the UK.

However, the major limitation of the study is that no quantitative methods for measuring effectiveness in terms of engagement or message diffusion were conducted, although the avoidance of Day Zero in particular offers some scope as a generalised proxy indicator. Assessing effectiveness is a key part of future research to determine which sub-categories of channels, content and messages were most effective.

Other studies of social media question the long-term benefits, particularly in terms of the ability of social media to change the nature of the discourse. In their study of water conservation messages and narratives on Twitter, (Boyer et al., 2021) found the platform useful to convey a standardized water conservation message as part of developing a wider discourse at global level. Amongst a network of water professionals and a community of water activists, the interactive element of Twitter helped to develop an institutional consensus. However, the authors found that social media was largely reinforcing of existing dialogues and narratives among water professionals and rarely challenged current water governance paradigms. They describe the result as a 'low level of dialogue intensity' where water issues are largely depoliticized amongst a background of maintaining institutional consensus. They conclude that Twitter's communication of water conservation agendas almost inevitably only offers incremental change. The implications for the purpose of messaging are profound.

'By insisting on the responsibility of individuals and inviting them to change their practices, the environmental marketing campaign de facto masks the political choices that lead to the construction of water facilities and spares large industrial and agricultural producers who consume a great deal of water and thus shifts the responsibility from decision-makers onto more diffuse culprits' (Boyer et al., 2021, p. 296).

In essence, responsibility is shifted to individuals and households, urging them to save water in order for the current water governance system to be able to continue.

This is a critique specifically of social media messaging, but as noted earlier, also extends to, and has implications for, all messaging about drought and more widely, water resource managing. In essence, messaging which implicitly or explicitly reinforces the status quo of industrial scale water use and risks to water quality and quantity, but requires individuals to change could be accused of 'missing the point' and, at worst, could be considered unethical or duplicitous. The implications for trust and ethics are noted below.

The various research findings suggest that social media – although much vaunted for its speed and usability (at least for some populations) - is not the magic answer for all public messaging, especially where more fundamental changes in narrative are required.

5.9.4. Storying

Rather than simply presenting bald 'facts' as an incentive for behaviour change, storying is a means of embedding information and developing ideas about possibilities of change in narrative forms. Although the purpose of storying varies widely – it can be one or more of imparting information, inviting reflection, providing comparisons, introducing new ideas or alternative behaviours - the narrative format aims to contextualise the message and provide a meaningful account for the audience.

Storying can be general – referring to groups or communities such as 'farmers in SE England' or it can personal such as 'Farmer Smith in Aylesbury'. The latter is more likely to be of interest to farmers in that particular geographic area. If the storying is further personalised, such as 'Farmer Smith in Aylesbury who is a diary farmer with 300 head of cattle' then the story becomes especially relevant for other diary farmers, but less so for arable farmers. Similarly, storying about an urban family living in rented accommodation in a flat in Cardiff is likely to be of interest to other urban dwellers, but of less resonance to people living in small hamlets in rural Norfolk.

However, storying can be used to establish understanding of these connections and make them more 'visible' as a way to influence behaviours and practices. For example, recent research explored how personalization in news stories about UK farmers influenced the intentions of general readers to help with drought (Wald et al., 2020). They find that personalized news stories are more likely than non-personalized stories to increase readers' empathy and perceptions of others' (in this case farmers) plight. Messaging based on narrative engagement where the reader is 'transported into' the story was directly related to reader empathy and indirectly associated with intentions to donate to assist those suffering from drought.

This suggest that personalized stories designed to produce prosocial intentions may be more effective when they combine 'transportation' with perceived suffering and empathy. However, (Wald et al., 2020) found no direct relationship between personalization of narratives and behavioural intentions. This finding may be due to the emphasis on economic suffering of farmers within the news stories. Previous work in agro-environmental settings found that economic messages resonate less with public opinions than environmental messages (Peterson et al., 2019). In other words, readers care less about economic impacts on others compared to environmental impacts and associated messages.

The DRY project also explored the role of storying and future scenario building through collaboration between scientists and communities (Liguori et al., 2021). Incorporating aspects of citizen science, the findings suggest storytelling, rather than just being retrospective accounts of events, can be a creative process for imaging the future based on sound science and community aspirations. The scenario building aspects of storying may be a significant opportunity for exploring the range of possible behaviours and practices to explore how individuals or communities could act in any given type of imagined situation relating to drought.

The scope for messaging arising out of storytelling is an area for future research and could also be linked into developing context relevant social norms.

5.10. Trust

As noted above, on the most fundamental questions for any public messaging strategy is the extent to which the author(s) of the message is trusted by the public or intended audience. If the messenger is a water company then trust is likely to be dependent on public expectations about the water company meeting legal requirements on drinking water while prioritizing social and environmental goals in relation to profit motives. If a regulator, then trust will hinge on perceptions of the regulator's role and its performance. Similarly, trust in an NGO can depend on the extent to which the NGO's aims and role effect change – though the use of 'shock' tactics can backfire in terms of public support. In practice, there will be blurring across these boundaries and the author of a message may be judged differently in relation to different aspects of activities – i.e. water supply compared to sewage treatment.

Research in Australia found that where residents trust that water resources are being managed effectively then residents are more likely to conserve water generating a virtuous circle of practice. The reverse is also possible (Lowe, Lynch and Lowe, 2014).

In England and Wales, this is especially pertinent and significant in the context of recent increased concerns about river and bathing water quality. The role of water companies and the relationships between water companies and regulators has come under increased scrutiny due to public concerns about combined sewage overflows (CSOs) and ongoing inland and coastal water quality issues.

A 2023 study commissioned by Ofwat of public trust in water companies reveals a mixed picture. While there is public trust in drinking water quality and associated supply, there is substantial distrust relating to value for money; investment; protection of environment and biodiversity; and prevention of sewage in rivers and seas as shown in Figure 2.

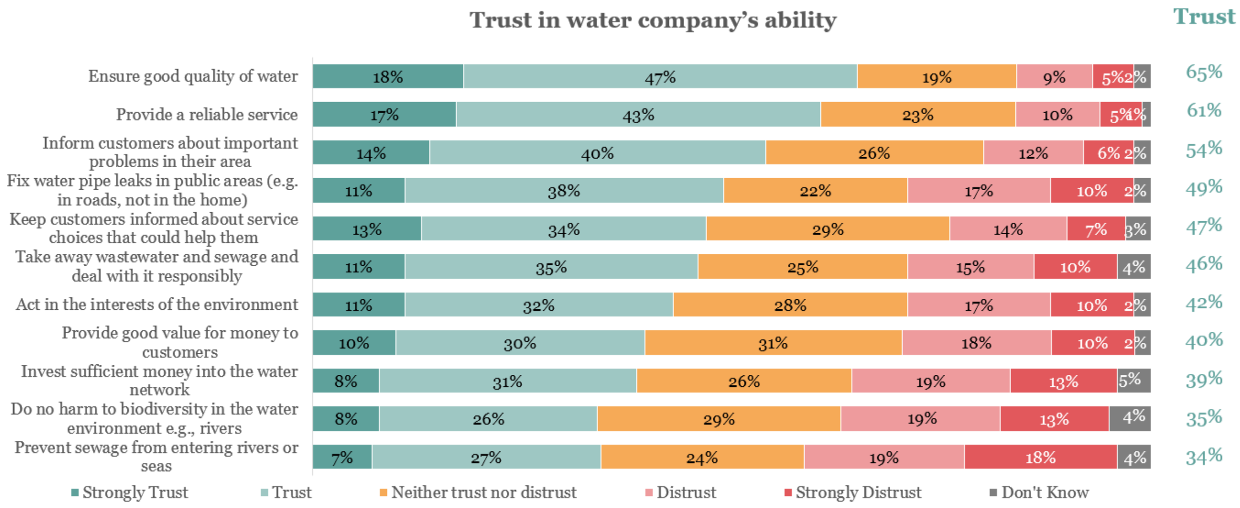


Fig 2 Trust in water companies' abilities (Ofwat, 2023)

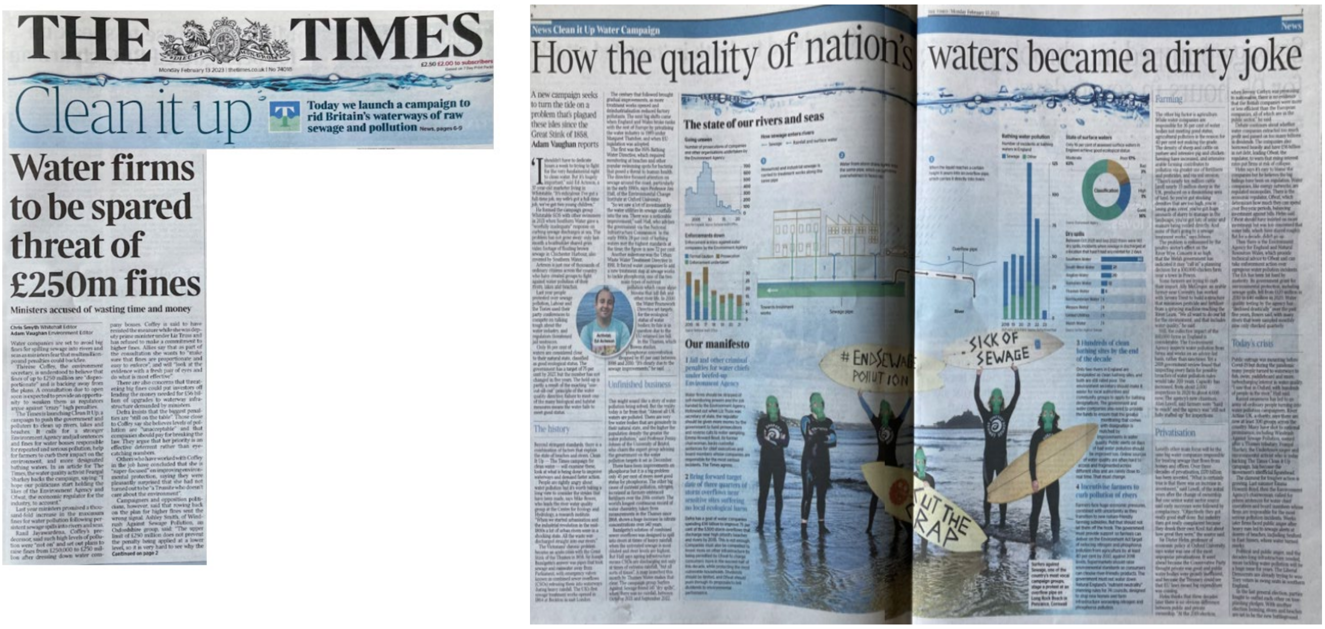
Figure 2 also suggests that focusing on drought as a quantity issue is only addressing part of the wider system of water which also includes water quality, environmental protection, organizational roles, regulatory practices, social values and expectations and so on. In other words, the public do not separate out the individual message from the context in which they perceive it exists.

5.11. Competing messaging

A focus on trust in relation to public messaging quickly opens up wider debates about the governance of water more generally. In addition to use by various stakeholders and different sectors, this might also include ownership of water companies, investment programmes, shareholder returns, executive renumeration, performance, quality standards, the role of Ofwat, DEFRA and the Environment Agency.

While the aim of public messaging is to change behaviours and practices, much of the literature on public messaging has tended to focus on 'how' and the different options associated with a framing of improving efficiency. There has been much less attention on the more fundamental question 'why are we managing water this way?'.

As noted above, social media tends to reinforce existing paradigms and narratives, but in the UK more recent media commentary has become much more critical of current water governance as shown in Figure 3. In many respects, this constitutes an alternative and competing messaging about water governance and could overshadow or undermine more 'conventional' drought messaging from water companies and similar sources.



THE TIMES

Clean it up

Today we launch a campaign to rid Britain's waterways of raw sewage and pollution News, pages 6-9

Water firms to be spared threat of £250m fines

Ministers accused of wasting time and money

How the quality of nation's waters became a dirty joke

The state of our rivers and seas

Our manifesto

Figure 3 Media Reporting in The Times newspaper, 13 February, 2023

The media critique can be summarised by questions such as 'In the 30 years since privatisation, why are CSOs still an accepted and permitted feature of water governance in England and Wales?' and 'Is current water governance privatising profit and nationalising pollution?'. These questions may seem overly political and somewhat removed from immediate drought concerns and the specifics of public messaging, but they are essential to understanding the context of drought messaging and the extent to which trust is linked to the potential for changed behaviours. In other words, the media critique is offering the public a competing message which aims to reframe ideas about what constitutes socially and environmentally acceptable water governance. In this context, continuing with 'public messaging' which continues to advance the status quo is unlikely to be effective.

This is not to suggest that current critiques are 'correct', but public messaging about drought which fails to take account of a growing public discontent about environmental and water quality and which simply reinforces (even by omission) existing governance arrangements and narratives, is less likely to be trusted and acted upon.

In other words, and bluntly, there is likely to be limited public support for and significant reputational risk to a water company urging and expecting domestic users to stop, for example, using a hosepipe during a drought when the water company is polluting a river, paying shareholders significant dividends and CEOs substantial renumeration. The reputational risk and loss of trust in the regulatory organisations is likely to be proportional to the extent they are permitting (figuratively and literally) such a situation to continue.

Inviting other, more trusted organisations such as an NGO to ‘do the drought messaging’ may be an option or, as the South African experience showed, combining messages on a shared platform. But NGOs may well be similarly at risk of reputational damage and at risk of being compromised if drought messaging is divorced from wider debates about water governance.

6. Concluding comments

6.1. Key social and behavioural scientific insights about public messaging on drought

There is no single understanding of drought nor a single homogenous ‘public’ for which a unified model of drought and drought related messaging is universally relevant and applicable.

Public messaging is a very broad term that usually refers to content being made available to members of the public which may or may not carry expectations of behaviour change. It is less commonly understood as the wider process or system of communication and learning about drought in the public sphere. In its more common interpretation, public messaging can take many forms from one-off social media tweets and posts, emails, printed information leaflets and signage, to organised events, comprehensive publicity and media campaigns and associated reporting.

The purpose of public messaging is often assumed to be to change public (usually household) behaviours in order to achieve or progress some measure of sustainability. But this purpose is framed from the originator’s point of view and may be received and experienced as something more negative such as ‘interference’ or ‘greenwash’.

The prevailing assumption that improved public communication or messaging leads to positive shifts in behaviours in regard to water use is now widely questioned. In turn this opens up to scrutiny the purpose of messaging. Is it to encourage efficiency type behaviours within the existing water system or is it aiming to fundamentally question existing practices and substantially transform the way we think about water? Messaging in either case will be designed, communicated and received very differently.

Messaging to ‘correct’ unsustainable behaviours carries several assumptions including what constitutes sustainable is known; individuals are equally responsible across all parts of society and those seeking change know the right course of action. Messaging framed by these assumptions implies judgement, failure and guilt by individuals (compared to organisations) as well as a superior understanding of the situation by others and knowledge of the optimal action to target.

Questioning the purpose and underlying assumptions of public messaging gives rise to questions about current understanding of behaviours and appropriate messaging.

Human behaviour can be defined as the potential or expressed capacity of individuals to respond mentally, physically and socially to internal and external stimuli. Internal factors (beliefs, values, attitudes, emotions, and knowledge) and external factors (contexts, formal regulation, social and cultural norms) are all important considerations in behaviour change strategies.

While many models of human behaviour can be employed, a broad distinction exists between focussing on behaviours and focussing on practices.

Behaviour change studies have tended to be couched in and focus on the individual framed as economically rational and therefore responsive to rational messaging, usually in the guise of a knowledge deficit model where messaging provides the 'facts' to reset behaviours.

More recent research moves away from the rational model of economic efficiency and maximisation of economic utility, towards a more complex understanding of behaviours where 'irrationality' or 'more than rational' decision making is accepted as a legitimate position and strategy. This leads to an understanding that behaviour is more often 'unconscious', energy-efficient, quick, and based on intuition and emotions, and individuals' lack of mental energy, time, and capacity. Multiple routes for making choices are therefore possible. This remains at odds with policy maker's expectations and persistent assumptions of public behaviours headlined by recourse to 'common sense' and the importance of 'getting the right message across'.

Recognising the complexity of behaviour has led to an arguably more systemic understanding and approach to behaviour focussed on practice - routine behaviour arising from multiple interconnected elements and things.

Messaging which focusses on behaviour change to reduce water consumption does not automatically change the context in which water is used; risks ignoring the variety of 'normal' consumption practices and ways of living and is unlikely to have much effect.

Significantly, messaging within an efficiency paradigm reproduces specific understandings of 'service' which may not be sustainable in the long run and ignores the transformative value of water in individual livelihoods. Messaging which continues to endorse and 'allow' the current framing of water use is at risk of missing the point.

Nonetheless, many barriers to behaviour change remain including individual attitudes and temperament; sense of responsibility, trust and efficacy; and practicality including lack of time and resources.

In general terms, messaging for behaviour change programmes should therefore focus on selecting the key behaviour to realise the largest change (this may not be the individual) and choose approaches based on context and objectives. Control groups may or may not be appropriate, but pilot studies are advised with clear definition of measures of success and methods for assessing defined outcomes. Monitoring is essential, especially long run monitoring to determine impacts of the measures and provide new interventions as needed.

However, behaviour change is just one of several causes of environmental degradation and over-emphasising behaviour change when other socio-institutional barriers are dominant may be ineffective.

Even so, accurate predictions of how individual people will behave in any given situation therefore remain elusive. A key barrier is that policy-makers continue to rely on platitudes about ‘getting people to change their behaviour’. This ignores behaviour bias including a tendency for simplification of future risks to zero; reliance on available information as fact; the finite pool of worry; myopia or short-termism; and herding.

A range of suggested behaviour change tactics and strategies are evident in the literature, but there is limited quantification of impacts. The strategies can be divided into the following broad categories:

1. education and awareness
2. outreach and relationship building
3. social influence
4. nudges and behavioural insights
5. incentives
6. storying.

There is considerable variation within each category and therefore advantages and disadvantages depending on context and purpose. Specific suggestions are set out in 6.2 below.

Whichever strategy is adopted, several studies show that trust in the author of the message is central to a positive reception. Current media and wider concerns about water quality in rivers and coastal waters in England and Wales are especially relevant and there is some evidence that trust in water companies has decreased for certain activities. This extends across the water governance system to include regulators. Where water companies remain in public ownership such as Welsh Water, trust is likely to be higher providing a more conducive foundation for behaviour change.

Public drought messaging does not happen in a water governance vacuum and will be assessed accordingly to the extent that other water issues are addressed. In many respects, the media concerns about water quality constitute a competing message about water governance, namely: 'why are we managing water this way?'. There is a very real risk that public messaging is used as a means to enable the status quo to continue where the status quo is considered by the public to be less desirable than other possible alternatives.

6.2. Advice on the timing, content and how messages should be delivered

While there are a number of recent reviews relating to behaviours and messaging, evidence for changes arising from interventions is patchy.

Most studies are of limited time span (<24 months) and therefore timing is not always explicit in the literature. However, there is recognition that 'early' interventions such as warnings need to be followed up and reinforced with 'later' messaging partly to contextualise earlier messaging, but also because of behaviour 'fade' and even the possibility of rebound effects. Over time, the messaging cycle should be characterised by 'long-term' actions punctuated and augmented by 'short-term' more immediate actions as needed within drought.

However, there is a delicate balance to be struck between the dynamics of too much or too little messaging and content. Background updates and ongoing communications are relevant strategies but in an increasingly online world, information overload is a risk. It is the case that most individuals, particularly in urban contexts, do not know or care about where their water comes from or goes to in general terms, though that may change rapidly with awareness of local impacts (such as sewage spills, stream drying, and floods or other indirect impacts relating to energy or food.)

Forecasting drought onset, duration, intensity and end remains an imprecise science. Ongoing 'background' messaging needs to be honest about this uncertainty and begin more a focus on resilience and building adaptive capacity for managing water as a totality comprising supply, quality, and droughts and floods as relevant to specific localities.

Pre and in drought messaging needs to be equally honest about the causes of drought – i.e. not just a function of rainfall but with causal human agency and how different sectors are preparing and responding to the same 'event'. This should not be a recourse to the bland implication of being 'all together in the same boat' since this is clearly not the case with some individuals' practices having limited capacity for change in the short term compared to others. This also extends to different organisations and sectors.

Content of messaging is very difficult to determine and specify without reference to particular contexts, audiences and the barriers to change and particular bias any audience segment might experience. There is a trap in thinking that a single message will achieve change when behaviours understood as practices are mediated by localities (urban/rural), histories, households, technologies, cultures, social status, ethnicity, education, income, previous experience of droughts or similar environmental event and so on. In addition there is a trap that behaviour change is the answer whereas a focus on structural issues and policy may be more appropriate to bring about situation improvement.

To address some of these issues, evidence from the literature suggests that – where appropriate -messaging must be flexible and multi-layered in terms of content and over time to bring about short-term behaviour shifts and longer-term new habits in relation to droughts. A range of tactics are available across a spectrum including knowledge transfer through to incentives. But the literature is also clear that identifying a specific behaviour requires careful analysis whether the desired change will deliver the expected benefits to the drought situation. A key message from the literature is that messaging must enable positive action (efficacy) – i.e. an individual is able to enact some kind of change on the basis of the message. Thus, messaging must take into account what can be done in urban compared to rural contexts and for different demographics.

The format and delivery of messages is key. Posters and information leaflets have minimal impact except for those predisposed to pro-environmental behaviours. Social media while relevant for short term messages, is also not a panacea because it is not ubiquitous and reliance on it risks reinforcing digital divides. It is also unable to move much beyond one-way communication. It has limited scope for reshaping discourses and narratives, instead reinforcing existing traditions, institutions, and arrangements of water governance. But it can be useful for certain audiences and existing networks to distribute information, though caution must be exercised if celebrity endorsement of particular messaging is considered.

It is appropriate that some messaging can be done by water companies or handled by marketing specialists employed by water companies – albeit at the risk of trying ‘to sell’ behaviour change and generating more top-down messaging via emails, leaflets and similar. While this can be successful in the short term, particularly a crisis, longer term behaviour changes are unlikely.

Evidence suggests that messaging from a collective platform of stakeholders (rather than an individual company or organisation) is more likely to be trusted and acted upon, though caution must be exercised by ‘farming out’ messaging to, for example, a respected NGO. Delivery through existing networks of family and friends/neighbours offers some scope for experimentation and lays a foundation for citizen science.

But as a form of communication, the term “messaging” itself carries an association with a short format and implies being passed from one to another in the sense of “pass the message along”. Synonymous with knowledge transfer and the knowledge deficit model, it implies a one-way direction and a linear sense of process such that the original content (decided by whom?) remains unchanged at the point of being received and acted upon.

The extent to which messaging is framed in this way will shape the design and content of messaging and ultimately its potential for behaviour change. Conceptualising messaging as a two-way or even multi-actor/node process offers more scope for considering messaging as (part of) a process or system of ‘knowing about and acting about drought’ where the emphasis is on an ongoing interchange between those creating and receiving messages. In this conceptualisation, messaging created by users and messaged to organisations and vice versa is understood as a dynamic learning system rather than a one-way knowledge exchange (often singular) event.

In other words, there is scope to imagine a more systemic rather than deterministic view of the relationship between the message, messenger and messaged. In the context of citizen science, this would allow for the ‘public’ to generate its own messaging for itself and to inform other members of the public and organisations and to feed this back into the water companies and regulatory organisations.

Involving citizens and users in the discussion will shed light on the bigger question: what is the narrative we need about water in the UK in a climate changing world and where/how does drought fit into this narrative and how will individual behaviours – alongside changes by other industrial users - contribute to adaptation and amelioration?

The public are being encouraged to adapt to drought / climate change, but this is to miss our own (human/ societal) agency in creating climate change and exacerbating drought and floods in parallel with water quality issues.

6.3. A view on the limitations of the evidence base and recommendations for improvement.

The evidence base for public messaging is rich, diverse and difficult to pin down with certainty for any specific locality / situation / time. The evidence base is also very patchy in terms of quantitative studies, but is considerable in terms of examples of different messaging strategies. But these are largely short term and often rather siloed explorations of how a household uses a particular resource at a particular time.

A more systemic approach to future studies on public messaging would be first to research how an individual / household understands what it means to be more sustainable in terms of water: what is easy and can be prioritised and what is difficult and not priority. A focus on practices offers considerable scope for situating messaging in the realities of peoples’ lives rather than abstract exhortations. This provides the basis for expanding research into messaging to existing networks of friends and neighbourhoods and citizen science initiatives.

7. What we don't know

Despite many studies, specific, quantifiable evidence of savings by households arising from different types of messaging and changed behaviours is still limited. Similarly, quantitative measurement of the impacts of messaging on behaviour is still ad hoc. This is a difficult area to research over the long term for a range of technological and social reasons especially attributing effectiveness to particular messages when behaviours are always highly contextual.

Outside of specific studies, current estimates are mostly linked to supply company estimates and volumes rather than actual measurements of individuals and households. There is also limited insight into whether behaviours and practices have changed directly or indirectly as a result of messaging and which type of messaging is the most effective in any given context.

There is no easy answer to address this gap which is also shaped by a prevailing ethic of human right to water and concerns over water company profits, behaviour fade, water meters in only half of UK households and the negligible use of smart water metering.

We know that public understanding of drinking water supply and sewage systems is patchy. But there is limited research about public understanding of droughts as part of seasonal and climate-related dynamic of water. Public understanding of human agency in exacerbating and / or causing droughts is also under-researched.

In addition to gender and social aspects of behaviours, there is limited understanding of messaging for often diverse and multi-cultural urban population where water is largely 'hidden' either physically or socially and culturally. Research is needed on the ways in which ethnic cultures within the UK value, understand and/or respond to environmental concerns generally, water specifically and drought in particular.

Regulators, water companies and water governance systems are under increasing scrutiny in the UK due to river quality and ongoing debate about profits, ownership and investment. The extent to which this debate erodes public trust and willingness to change behaviour is unknown, but a recent survey by Ofwat suggests public trust is reducing in terms of environmental protection of rivers and habitats. This finding ties into a much longer ongoing debate about coastal bathing water quality. Further research is required on whether levels of trust in water governing organisations are determining how drought messaging is received and acted upon or rejected.

Similarly, there is limited insight into possible rebound effects associated with adaptive behaviours such as increased water use arising from the notion that the water problem (however defined) is resolved due to new investments / storage / treatment capacities/demand efficiencies. With planned system infrastructure and UK interlinkages this could be a significant concern, especially if regional areas experience significant climate extremes. In other words, the possibility of mass transfers from ‘wet’ areas to SE England could cause water demand in SE England to increase.

8. What we should know

Whatever messaging strategy and format, there is limited evidence for longevity of the impacts of messaging on behaviours in the long-term, though some studies suggest that focussing on social norms does enable long term adaptations. However, there is almost no consideration of the welfare implications of changing behaviours and the social / psychological impacts on water users who feel under increasing social pressure to reduce consumption. More research is needed on the risks of disenfranchising or blaming certain groups of individuals and sectors who, in their view, may have valid reasons for using water in the way that they do. The dissonance of focussing on individual change against a backcloth of wider infrastructure and governance problems in water governance should be addressed in future research.

Nonetheless, social norms are powerful ‘carriers’ of messaging. While intra-group (you and your neighbours) comparison is already known to be of value, future research should also further consider intergroup comparisons (urban compared to rural) as a way of motivating behaviour change, although there is a known risk of undermining sustainable actions where groups ‘give up’ following their comparison with others requires additional research.

Barriers to behaviour change are relatively well understood, but will continue to change over time in scale, scope and form. Messaging must be receptive to this. Current financial concerns for many households may mean financial incentives to reduce water demand could have more traction especially if linked to environmental quality. The scope for reducing water consumption for financial and environmental benefits could be an area of further research, but the divide between metered and unmetered users is likely to be a practical problem in pursuing incentives based on use reduction. Moreover, rightly or wrongly, water as a human right is a prevailing ethic, paralleled by a sense that water supply should not be a profit-orientated industry.

Most of the debates about current messaging are about aiming for efficiency and saving water in response to droughts framed as an external event, rather than a larger narrative about how human activity is a causal factor in the availability of water. This larger framing could be key to encouraging new understandings, a new sense of agency and new behaviours where human practice is increasingly recognised as a causal factor in drought onset and managing. The extent to which this wider narrative of causal agency is meaningful and empowering for the public or experienced as demotivating as just more ‘guilty responsibility’ requires further research.

Recognising causal agency in drought brings with it a requirement for a more systemic understanding of drought and human behaviour and practice. This means exploring how public messaging about drought should engage with the interdependencies in the water system and the diverse actors and the emergence of drought. While this will be challenging to convey – at least in short messaging formats - at the very least, more consideration is needed how to convey feedback loops to show relationships between an individual’s pro-environmental actions and environmental benefits. Currently much messaging is about ‘do X to save water and / or money / or help the environment’ but less about ‘the savings you [collectively] made have meant river or water table levels remain stable and abstraction has not needed to increase’. Of course, this is a simplistic example, and there are many complexities about making and evidencing claims for causal relationships and feedback loops. But the type and effectiveness of messaging where this may be possible needs to be understood. The role of simple diagrams and infographics to convey feedback and interdependencies will also need to be considered.

Climate change is likely to bring increased pressure on improving water governance for both drought and flood extremes, although occurring at different intervals and intensity. To adapt and build resilience will require a step change in our collective understanding of water in the UK and will prompt fundamental questions such as ‘why are we doing water governance this way?’. The recent widespread media coverage of sewage releases into rivers is but one element of this wider conversation about the future of water in the UK. Public messaging will need to be developed which acknowledges and responds to ‘water governance in a climate changing world’. Messaging on this wider narrative is essential to begin building deeper conversations about how households can contribute to the ways water now needs to be managed. Messaging to ‘turn off the tap’ or ‘use a watering can’ while relevant, is no longer sufficient in a climate changing world.

Against a background of an increasingly urban population where water and droughts are often ‘invisible’, further work is needed to understand what public messaging would look like when combined with science to reflect and acknowledge the narrative elements of droughts as understood and experienced by different stakeholders. This could further develop the social norms of messaging which the above review suggests has a positive effect on behaviours.

This leads to perhaps the most important requirement for future research. Currently, most messaging is largely one-way: from organisations to the public. The motivations, capacities, mechanisms, skills and languages to enable two-way messaging, learning and ultimately, new practices, require further investigation. The growing interest in more communicative processes such as citizen science and living labs could provide a fertile research area for understanding what two-way communication and two way messaging could look like to address the complexities of water governing and drought managing in a climate changing world.

Currently, citizen science is an increasingly common part of water governance, especially in individual catchments where catchment groups are exploring a range of approaches to improve water quality. The scope for citizen science projects relating to drought planning and drought messaging is largely unknown, but offers potential for contextualising messaging content and related behaviours and offering meaningful localised actions – an essential aspect of messaging. It may also contribute to understanding drought as part of the water dynamic and the flood – drought spectrum. Research is needed to understand the extent to which citizen science could provide a more trusted element to two-way drought messaging and provide necessary context for communities (however defined) to alter practices.

While water companies may understand customers, it is currently uncertain as to the extent they understand people, households and communities situated in context.

References

Adams, D.C. et al. (2013) 'The Influence of Water Attitudes, Perceptions, and Learning Preferences on Water-Conserving Actions', Natural Sciences Education, 42(1), pp. 114–122. Available at: https://doi.org/10.4195/nse.2012.0027

Agachi E, Bijmolt THA, van Ittersum K, Mierau JO. (2023) The Effect of Periodic Email Prompts on Participant Engagement With a Behavior Change mHealth App: Longitudinal Study. JMIR Mhealth Uhealth. May 11;11:e43033. doi: 10.2196/43033. PMID: 37166974; PMCID: PMC10214118.

Agarwal, S., Araral, E., Fan, M. et al. (2022) Water conservation through plumbing and nudging. Nat Hum Behav 6, 858–867 (2022). https://doi.org/10.1038/s41562-022-01320-y

Ajzen, I. (1991) The theory of planned behavior, Organ. Behav. Hum. Decis. Process., 50, 179–211, https://doi.org/10.1016/0749- 5978(91)90020-T

Antwi, S.H. et al. (2022) 'Communicating water availability to improve awareness and implementation of water conservation: A study of the 2018 and 2020 drought events in the Republic of Ireland', Science of The Total Environment, 807, p. 150865. Available at: https://doi.org/10.1016/J.SCITOTENV.2021.150865

Asprilla Echeverría, J.M. (2020) 'Cross-country evidence for social dimensions of urban water consumption during droughts', Journal of Cleaner Production. Elsevier, p. 120895. Available at: https://doi.org/10.1016/j.jclepro.2020.120895

Badullovich, N., Grant, W.J. and Colvin, R.M. (2020) 'Framing climate change for effective communication: A systematic map', Environmental Research Letters, 15(12). Available at: https://doi.org/10.1088/1748-9326/ABA4C7

Berger, J. (2021) 'Social tipping interventions can promote the diffusion or decay of sustainable consumption norms in the field. Evidence from a quasi-experimental intervention study', Sustainability (Switzerland), 13(6). Available at: https://doi.org/10.3390/su13063529

Blair, P. and Buytaert, W. (2016) 'Socio-hydrological modelling: A review asking "why, what and how?"', Hydrology and Earth System Sciences, 20(1), pp. 443–478. Available at: https://doi.org/10.5194/HESS-20-443-2016

Botzen, W., Duijndam, S. and van Beukering, P. (2021) 'Lessons for climate policy from behavioral biases towards COVID-19 and climate change risks', World Development, 137. Available at: https://doi.org/10.1016/J.WORLDDEV.2020.105214

Boyer, A.L. et al. (2021) ‘Building Consensus? The Production of a Water Conservation Discourse Through Twitter: The Water use it Wisely Campaign in Arizona’, Environmental Communication, 15(3), pp. 285–300. Available at: https://doi.org/10.1080/17524032.2020.1821743

Chappells, H., Medd, W. and Shove, E. (2011) ‘Disruption and change: Drought and the inconspicuous dynamics of garden lives’, Social and Cultural Geography, 12(7), pp. 743–756. Available at: https://doi.org/10.1080/14649365.2011.609944

Dean, A.J. et al. (2016) ‘Fostering water sensitive citizenship - Community profiles of engagement in water-related issues’, Environmental Science and Policy, 55, pp. 238–247. Available at: https://doi.org/10.1016/J.ENVSCI.2015.10.016

Duthie, E. et al. (2017) ‘The effectiveness of celebrities in conservation marketing’, PLoS ONE, 12(7). Available at: https://doi.org/10.1371/journal.pone.0180027

Ferraro, P.J., Miranda, J.J. and Price, M.K. (2011) ‘The persistence of treatment effects with norm-based policy instruments: Evidence from a randomized environmental policy experiment’, in American Economic Review, pp. 318–322. Available at: https://doi.org/10.1257/aer.101.3.318

de Freitas Netto, S.V. et al. (2020) ‘Concepts and forms of greenwashing: a systematic review’, Environmental Sciences Europe, 32(1). Available at: https://doi.org/10.1186/s12302-020-0300-3

Ghoreishi, M., Razavi, S. and Elshorbagy, A. (2021) ‘Understanding human adaptation to drought: agent-based agricultural water demand modeling in the Bow River Basin, Canada’, Hydrological Sciences Journal, 66(3), pp. 389–407. Available at: https://doi.org/10.1080/02626667.2021.1873344

Hart, P.S. and Nisbet, E.C. (2011) ‘Boomerang Effects in Science Communication’, http://dx.doi.org/10.1177/0093650211416646, 39(6), pp. 701–723. Available at: https://doi.org/10.1177/0093650211416646

Hallaj, Z. et al. (2021) ‘Human ecological analysis of farmers’ pro-environmental behaviour in the face of drought: Application of Norm Activation Theory’, Water and Environment Journal, 35(4), pp. 1337–1347. Available at: https://doi.org/10.1111/WEJ.12733.

Kahneman, D., (2003) Maps of bounded rationality: psychology for behavioral economics. Am. Econ. Rev. 93, 1449–1475.

Katz, D. et al. (2016) ‘Evaluating the effectiveness of a water conservation campaign: Combining experimental and field methods’, Journal of Environmental Management, 180, pp. 335–343. Available at: https://doi.org/10.1016/j.jenvman.2016.05.049.

Kelly, M.P. and Barker, M. (2016) 'Why is changing health-related behaviour so difficult?', Public Health, 136, pp. 109–116. Available at: https://doi.org/10.1016/J.PUHE.2016.03.030.

Koop, S.H.A., Van Dorssen, A.J. and Brouwer, S. (2019) 'Enhancing domestic water conservation behaviour: A review of empirical studies on influencing tactics', Journal of Environmental Management, 247, pp. 867–876. Available at: https://doi.org/10.1016/J.JENVMAN.2019.06.126.

Krawczyk, K. et al. (2021) 'Quantifying online news media coverage of the COVID-19 pandemic: Text mining study and resource', Journal of Medical Internet Research, 23(6). Available at: https://doi.org/10.2196/28253.

Kuyer, P. and Gordijn, B. (2023) 'Nudge in perspective: A systematic literature review on the ethical issues with nudging', Rationality and Society, 35(2), pp. 191–230. Available at: https://doi.org/10.1177/10434631231155005.

Lamm, A.J. et al. (2018) 'Informing water-saving communication in the United States using the situational theory of problem solving', Landscape and Urban Planning, 180, pp. 217–222. Available at: https://doi.org/10.1016/J.LANDURBPLAN.2018.09.006.

Larbey, R. and Weitkamp, E. (2020) 'Water Scarcity Communication in the UK: Learning From Water Company Communications Following the 2018 Heatwave', Frontiers in Environmental Science, 8, p. 167. Available at: https://doi.org/10.3389/FENVS.2020.578423/BIBTEX.

Lede, E., Meleady, R. and Seger, C.R. (2019) 'Optimizing the influence of social norms interventions: Applying social identity insights to motivate residential water conservation', Journal of Environmental Psychology, 62, pp. 105–114. Available at: https://doi.org/10.1016/J.JENVP.2019.02.011.

Liguori, A. et al. (2021) 'Towards "Creative Participatory Science": Exploring Future Scenarios Through Specialist Drought Science and Community Storytelling', Frontiers in Environmental Science, 8. Available at: https://doi.org/10.3389/FENVS.2020.589856/FULL.

Lindsay, J., Dean, A.J. and Supski, S. (2017) 'Responding to the Millennium drought: comparing domestic water cultures in three Australian cities', Regional Environmental Change, 17(2), pp. 565–577. Available at: https://doi.org/10.1007/s10113-016-1048-6.

Lowe, B., Lynch, D. and Lowe, J. (2014) 'The role and application of social marketing in managing water consumption: A case study', International Journal of Nonprofit and Voluntary Sector Marketing, 19(1), pp. 14–26. Available at: https://doi.org/10.1002/nvsm.1484.

Mcalpine, A., Sahar, A. and Pannocchia, D. (2021) ‘Abase and Abound: Social Media Campaign Strategies during the Cape Town Drought and UK Winter Floods’ in AQUA≈360: Water for All - Emerging Issues and Innovations, pp. 2–4. Available at: https://shura.shu.ac.uk/29176

McEwen, L. et al. (2021) ‘Science-Narrative Explorations of “Drought Thresholds” in the Maritime Eden Catchment, Scotland: Implications for Local Drought Risk Management’, Frontiers in Environmental Science, 9. Available at: https://doi.org/10.3389/FENVS.2021.589980

OFWAT (2023) Trust and perceptions: People’s views on the water sector. Full report. Available at: https://www.ofwat.gov.uk/publication/trust-and-perceptions-peoples-views-on-the-water-sector-full-report/ Accessed 19 April 2023.

Pawsey, N., Nayeem, T. and Huang, X. (2018) ‘Use of facebook to engage water customers: A comprehensive study of current U.K. and Australian practices and trends’, Journal of Environmental Management, 228, pp. 517–528. Available at: https://doi.org/10.1016/J.JENVMAN.2018.08.063.

Peterson, D. A. M. et al. (2019). Carbon or cash: evaluating the effectiveness of environmental and economic messages on attitudes about wind energy in the United States. Energy Res. Soc. Sci. 51, 119–128. doi: 10.1016/j.erss.2019.01.007

Reckwitz, A. (2002) ‘Toward a Theory of Social Practices: A Development in Culturalist Theorizing’, European Journal of Social Theory, 5(2), pp. 243–263. Available at: https://doi.org/10.1177/13684310222225432.

Savelli, E. et al. (2022) ‘Drought and society: Scientific progress, blind spots, and future prospects’, Wiley Interdisciplinary Reviews: Climate Change, 13(3). Available at: https://doi.org/10.1002/WCC.761.

Schrieks, T. et al. (2021) ‘Integrating Behavioral Theories in Agent-Based Models for Agricultural Drought Risk Assessments’, Frontiers in Water, 3, p. 104. Available at: https://doi.org/10.3389/FRWA.2021.686329/BIBTEX.

Shove, E. (2018) ‘What is wrong with energy efficiency?’ Building Research & Information 46(7), pp. 779–789. Available at: https://doi.org/10.1080/09613218.2017.1361746.

Shove, E. (2014) ‘Putting practice into policy: reconfiguring questions of consumption and climate change’, Contemporary Social Science, 9(4), pp. 415–429. Available at: https://doi.org/10.1080/21582041.2012.692484.

Smith, K. et al. (2012) Reviewing Approaches for communicating Drought status And Risk (RADAR). Report to the Department of Environment, Food and Rural Affairs. Unpublished.

Tang, Z. et al. (2015) 'Examining the role of social media in California's drought risk management in 2014', Natural Hazards, 79(1), pp. 171–193. Available at: https://doi.org/10.1007/S11069-015-1835-2.

Tijs, M.S. et al. (2011) 'Does water context influence behaviour and attitudes to water conservation?', Australasian Journal of Environmental Management, 18(1), pp. 47–60. Available at: https://doi.org/10.1080/14486563.2011.566160.

Tortajada, C. and Nambiar, S. (2019) 'Communications on technological innovations: Potable water reuse', Water (Switzerland), 11(2). Available at: https://doi.org/10.3390/w11020251.

VanDyke, M.S. and King, A.J. (2020) 'Dialogic Communication Practices of Water District Officials: Insights from Practitioner Interviews', Environmental Communication, 14(2), pp. 147–154. Available at: https://doi.org/10.1080/17524032.2019.1705365.

Wald, D.M. et al. (2020) 'How Does Personalization in News Stories Influence Intentions to Help with Drought? Assessing the Influence of State Empathy and Its Antecedents', Frontiers in Communication, 5. Available at: https://doi.org/10.3389/FCOMM.2020.588978/FULL

Water UK (2023) Water Meters. Available at: https://www.water.org.uk/advice-for-customers/water-meters. Accessed 10 March 2023

Water UK (undated) Water – The Facts. Why Do We Need to Think About Water? Available at: https://www.waterwise.org.uk/wp-content/uploads/2018/02/Waterwise-2012_The-Facts_Why-do-we-need-to-think-about-Water.pdf. Accessed 20 May 2023

Whitehead, M., Jones, R. and Pykett, J. (2011) 'Governing irrationality, or a more than rational government? reflections on the rescientisation of decision making in british public policy', Environment and Planning A, 43(12), pp. 2819–2837. Available at: https://doi.org/10.1068/a43575.

L: Water Supply: Observed and Projected - Review of the state of research on drought

Chris Counsell, Mason Durant

HR Wallingford

Overview

THE REVIEW: Covers historical droughts, causes and variability of droughts, future droughts, present and future impacts and planning approaches. It summarises what we know, what we don't know and what we should know about water supply drought.

What is covered: Public water supply in England as well as other areas of the UK and internationally where relevant. Other users and uses of water more generally are discussed where there are implications for public water supply.

What is not covered: Short term operational drought planning and forecasting.

KEY FINDINGS:

What we know	What we don't know
✓ Water supply systems can generally be described as multi-season or single season vulnerable. These vulnerabilities tend to change over time (e.g. climate change increasing aridity, responses to drought events), can exacerbate drought conditions, and are still being revealed through recent droughts. ✓ Most water companies' worst historical droughts (events within the observed record that cause the most severe system response) are since 1920 and are generally three years or less in duration. ✓ There is an assumption of deep uncertainty for water resource planning. ✓ The current 1-in-500 plus climate change assessment likely misses variability, timing, duration, spatial extent and extremes of droughts within climate models. Assessment of future return periods and plausibility is almost impossible.	× Potentially more diverse events have occurred historically. The maximum severity, duration and spatial extent of drought events in the current and future climate is unknown, as well as the likelihood of dry, or successively dry winters in the future. × Uncertainties are not fully explored within current top-down methods, including land use changes, other sectors demands and future policy. Bottom-up approaches, better suited to deep uncertainty, are untested within the UK water resources decision context. × The full envelope of plausible demands and future climate change projections is unknown.

RECOMMENDATIONS:

- A **multi-sector approach, including concurrent and interacting risks** is missing, requiring understanding beyond system aggregating approaches such as deployable output. Improve understanding of demand changes due to behaviour, how demand reductions and system optimisation may impact vulnerability.
- **More diverse events** are required to examine system responses (particularly multi-season, **successive dry winter**, large spatial extent events). There is a need to **consider uncertainty** in historical and synthetic datasets.
- The specific requirements of UK water resource planning should be understood, so that **any move away from change factors represents an improvement** in the processes and variables of interest.
- **Process-based plausibility assessment should be used to assess climate projections**. Consideration should be given to how plausibility information can be applied to datasets and carried through to planning decisions.

1. Introduction

Water supply is a multi-faceted issue, incorporating physical, societal, economic, political and geographical aspects. Consequently, there is a large amount of literature covering a vast array of topics of relevance to water supply, including grey literature from applications in industry. The sometimes disparate nature of the literature means that it can be difficult to synthesise information across these different aspects. This thematic review pulls together the available literature on water supply, focussing on England, drawing on areas outside England where relevant, as well as overseas. The review focusses on public water supply due to the wide ranging supply network that provides water to the majority of users in England, however other users and uses of water more generally are discussed where there are implications for public water supply. Previous reviews covering water supply issues have tended to take a top-down approach, focussing on the climatology initially before discussing system implications. This review differs from previous literature in that it attempts to frame the literature through the lens of system vulnerabilities, providing a system-relevant perspective unique to the issues of water supply.

The first section focusses on historical water supply droughts, followed by a review of the causes and variability of droughts. Future water supply droughts are then discussed, before reviewing the impacts of water supply droughts in both the present and future. The review concludes with a section on current and future planning approaches. Each section is summarised by what we know, what we don't know and what we should know to improve our understanding.

2. Historical water supply droughts

There is a significant amount of literature covering historical water supply droughts, through reviews undertaken at the time of droughts (e.g. Doornkamp et al., 1980), retrospective reviews (e.g. Marsh et al., 2007; Rodda and Marsh, 2011; Durant, 2015; Water UK, 2016), and programmes that look to collate information across many aspects of historical droughts (e.g. UK Drought and Water Scarcity Programme – NERC Historic Droughts). This review does not attempt to revisit these, instead it focusses on the historical perspective through the lens of water supply system vulnerability. An overview of public water supply system vulnerability in England is presented, followed by a discussion of the historical perspective of drought timing, frequency, duration, severity and spatial extent. Short and long term responses to drought and the use of historical datasets are also discussed.

2.1. Public water supply system vulnerability

Public water supply systems at the water resource zone level in England are often complex, have evolved (and are continuing to evolve) over time and are unique when compared to one another. Consequently, the vulnerabilities of each system to the spatial pattern, severity, frequency and duration of drought events are different. Attempts have been made to generalise the response of water supply systems in the UK to different types of drought events, which has led to a general classification based on drought duration of vulnerability (Environment Agency, 2015) to short, single season (tending to be defined as up to 12-18 months in duration) or multi-season droughts (longer than 18 months). This subjective method of synthesis has been extended to water supply systems across England using drought response surfaces (where available) produced as part of Drought Plans and Water Resource Management Plans (WRMP) using the Drought Vulnerability Framework (UKWIR, 2017), and is presented in Figure 0.1. It is worth noting that the system definitions are not always clear, in many cases because there is a mixed, or complex response to a lack of rainfall in some systems. An example drought response surface is shown in Figure 0.2. Further examples can be found in (UKWIR, 2017).

Figure 0.1 shows that the majority of English water supply systems are multi-season vulnerable, with the largest single season systems tending to be located in the north and west of England. These systems tend to be single season due to a dominance of reservoir systems in catchments that have experienced regular winter rainfall where reservoirs refill quickly during winter months. A small number of single season systems are located in the south of England, where the chalk groundwater system is relatively flashy and responds quickly to winter recharge. Water resource zones in the south west of England that would be expected to be classified as single season systems due to their location, reliance on reservoirs and topography, but are multi-season systems, are largely due to the interconnected nature of the system, higher prevalence of groundwater sources (e.g. Wessex Water), or resilience schemes that alter the vulnerability of existing reservoirs (e.g. Wimbleball and Roadford reservoirs (South West and Bournemouth Water, 2022).

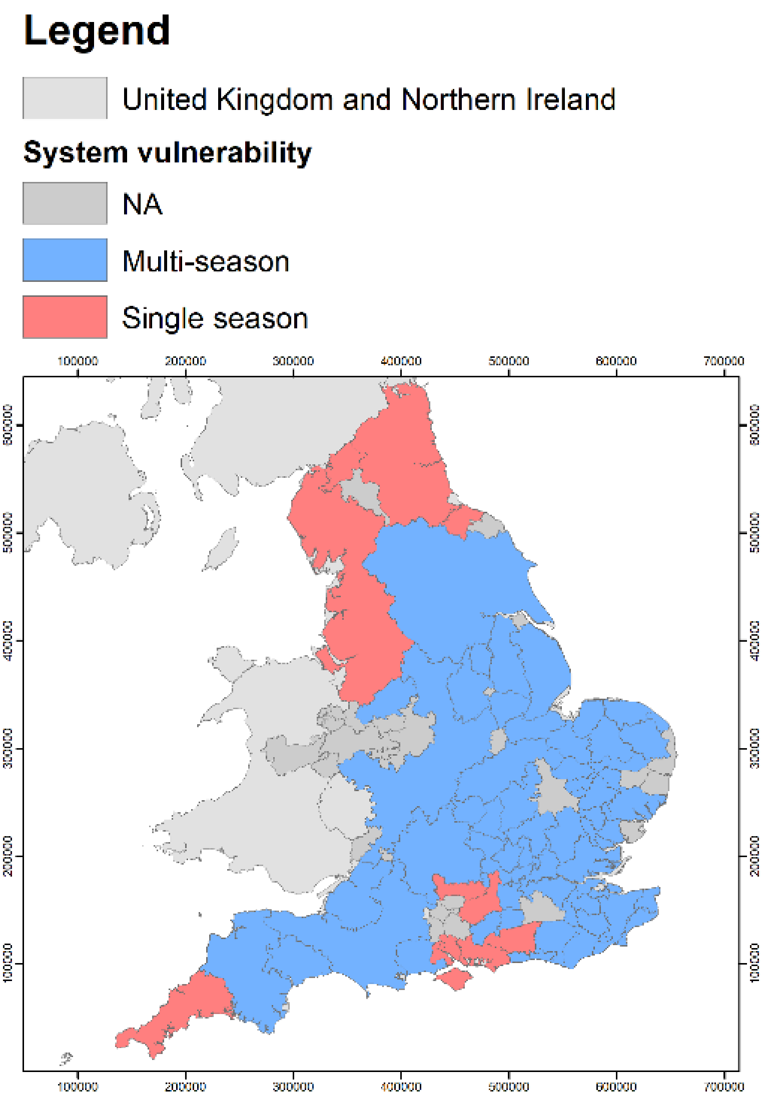


Figure 0.1: Approximated water resource zone vulnerability to multi-season and single season droughts, based on drought response surfaces contained within water company Drought Plans and WRMPs

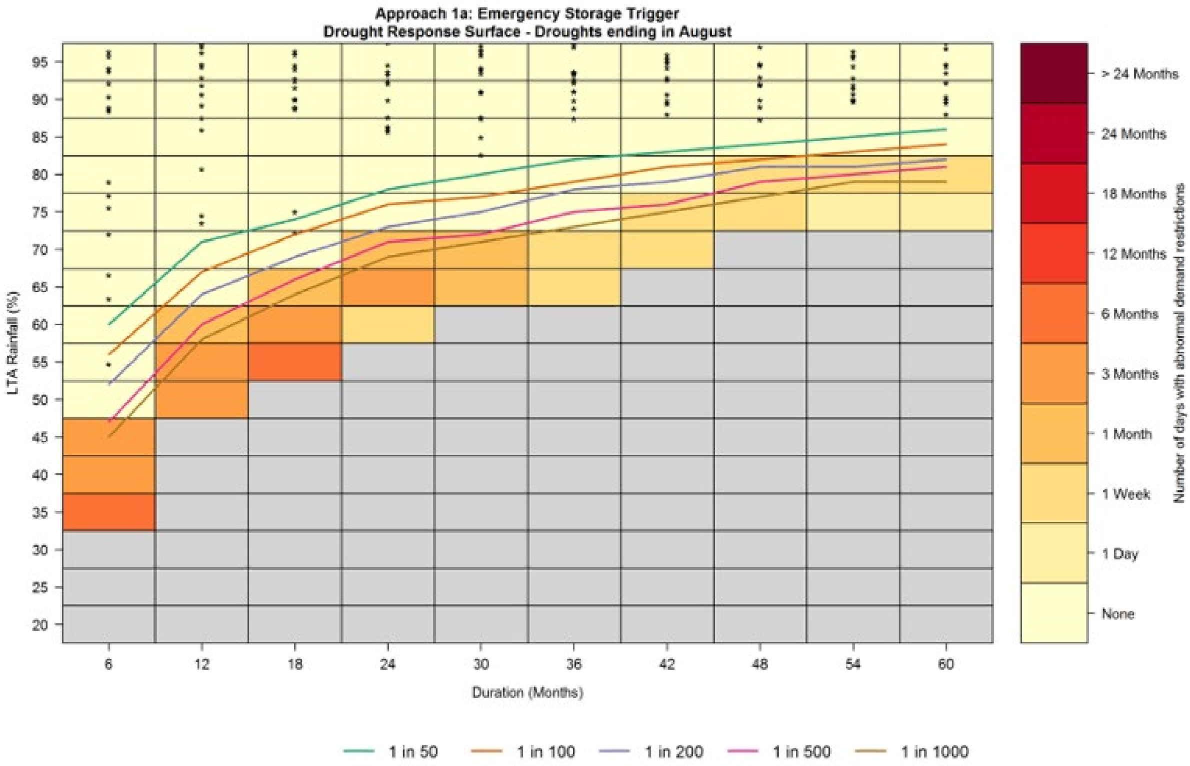


Figure 0.2: Example Drought Response Surface. Grey cells indicate unlikely rainfall totals which may be deemed "implausible" Source: UKWIR (2017)

2.2. Drought response

The human response to drought events via water supply system modelling can be considered in the short and long term. In the short term, responses to drought events are often undertaken in attempts to reduce demand or increase supply. For example, the implementation of Drought Orders and temporary use bans, publicity campaigns to reduce demand, the commissioning of new boreholes and river intakes, attempts to reduce leakage, temporary water transfers, pressure reductions and the installation of new booster pumps, and the restriction of spray irrigation (Doornkamp et al., 1980; Taylor et al., 2009; Rodda and Marsh, 2011; Durant, 2015; Durant and Counsell, 2020). The current approach within water company plans is that demand restrictions are incrementally increased as supplies run low, to the point where there are rota cuts and supplies are cut off (National Infrastructure Commission, 2018). This is unlikely to happen in reality, rather, emergency measures are undertaken to ensure supplies remain on (National Infrastructure Commission, 2018), as highlighted above. There are also lessons to be learnt from human responses in other areas of the world, such as Cape Town in South Africa in 2018, where water demand was reduced to just 50 litres per person per day as the drought progressed – an unprecedented and unexpected response achieved through an intensive public campaign (LaVanchy et al., 2019). Whilst English water companies prepare Drought Plans to enact responses during an emerging drought, the human response during an event is uncertain and can have an impact on the progression of a drought.

There is also evidence that experiences within drought periods shapes future management practices in the long term (Pearce, 1982; Miller and Yates, 2006; Taylor et al., 2009; Durant, 2015). Whilst frameworks exist to analyse the drivers, responses and impacts of droughts to find commonality and differences across different sectors, scales and through time (Lange et al., 2017), a systematic analysis of all historical events has not been undertaken. It is possible that doing so might shed light on these characteristics of drought, and therefore how populations might respond to different and future droughts. The number of variables and relatively small number of historical events would likely render results highly uncertain, however, it may still be worth undertaking such an assessment to improve understanding of human and community response. Emerging novel indicators using mining of public data such as Google Trends might provide insight related to trends in public awareness and interest in water saving measures during a drought event (Wilby et al., 2023).

2.3. Other users

The above sections focus mainly on the public part of water supply. However, public water supply systems are contained within a wider ecosystem of other users including agriculture, horticulture, industry and the environment. Some of these users are supplied directly through the public water supply system. These interactions have been part of English water resource management planning for some time. A number of other users are not directly part of, but do interact with, the public water supply system (such as those involved in agriculture that have private river abstractions). There is often a high degree of interdependence between other users and public water supply. Some of these users have made their own adaptations in response to historical events that could alter catchment resilience, but aren't included within water resource modelling (Rey *et al.*, 2017). Despite the knowledge that other users are impacted by droughts, there is a lack of systems literature on interrelations between these users. Multi-sectoral approaches that incorporate external water users are relatively new to water resource management planning in the UK and the resilience of other users and sectors to drought is not well understood (Environment Agency, 2020). The National Framework (Environment Agency, 2020) and regional planning (Environment Agency, 2021a) are attempting to involve these sectors where relevant, however little has been done to understand these interactions historically (Environment Agency, 2020).

2.4. Water demand

Water demand is a key component in historical water supply system drought. Demand is often defined as peak or annual, depending on the vulnerability of the system to either one of these types. Peak demands during a drought can cause supply issues, for example, in the summer of 1975 for some areas such as the Thames Valley, it lead to conveyance issues within the distribution network (Doornkamp et al., 1980). These issues continued into the autumn of 1975 and were repeated in the summer of 1976 in the Thames region (Doornkamp et al., 1980), West Midlands and Hampshire (Rodda and Marsh, 2011). Annual demands are often more of an issue over longer duration droughts for systems that are multi-year vulnerable. Water demand is discussed in more detail in the next section.

2.5. Drought characteristics

In addition to being able to elicit system vulnerability from the drought response surface, each response surface is produced on the basis of droughts ending in a particular month. That month will be considered the month that the system is most vulnerable to droughts extending into. Where available, this information has been extracted for each water company, as well as the duration and date of the worst historical droughts associated with each company, and is presented in **Error! Reference source not found.**.

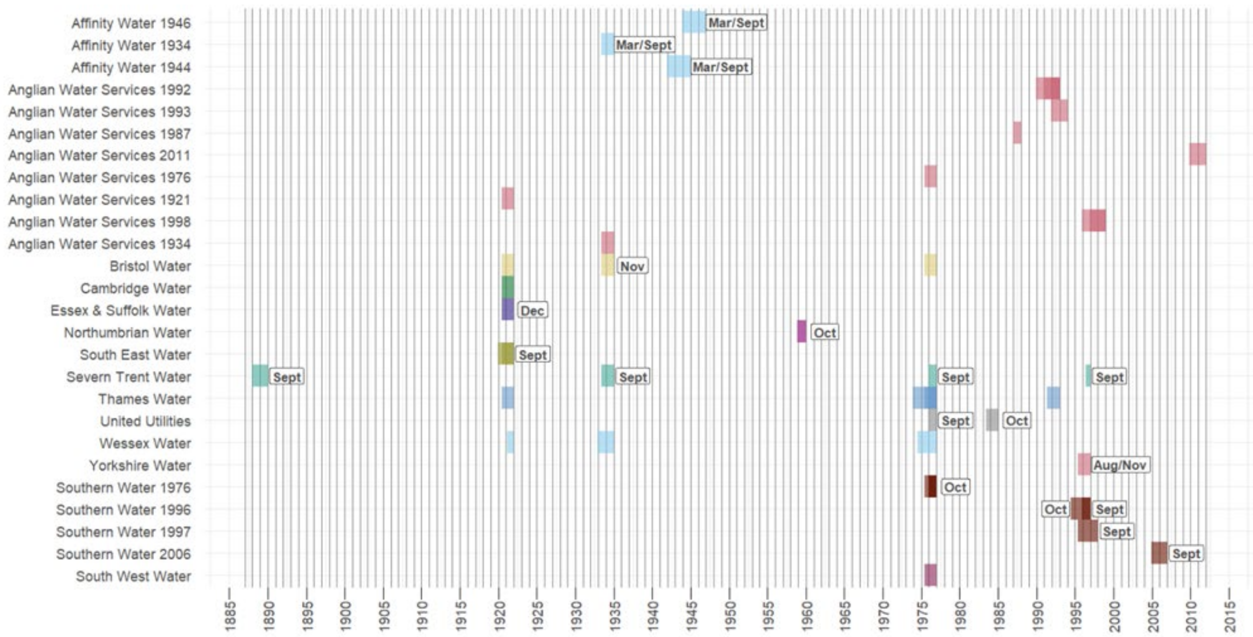


Figure 0.3: The worst historical droughts associated with each water company (extracted from recent Drought Plans and WRMPs) are shown where available. Where there are separate overlapping drought events that end in different years, historical droughts have been separated out for those water companies. The droughts are shown as ending in the December of the year of their termination, with durations estimated from information within the drought response surface, or literature in Drought and Water Resource Management Plans. Each vertical line represents a single year. It is worth noting that because the drought response surfaces generalise the system vulnerability, the month ending may not

correspond with the month the worst historical drought actually ended in. The month ending attribute instead describes the general vulnerability of the supply system

Error! Reference source not found. demonstrates that most water companies' worst historical droughts range from 1921 to 2011. Only Severn Trent Water (out of those companies analysed) have an historical event prior to 1921 that sufficiently tests their system to warrant inclusion. Whilst this does not mean that other companies haven't tested systems against droughts prior to 1921, this is generally the case as datasets with the required granularity of detail are not available prior to the start of the 19th century.

The month ending attribute of system vulnerability is mainly concentrated around September and October for those systems where data are available, indicating a significant sensitivity to autumn rainfall. A smaller number of systems are vulnerable to droughts ending in August, November and December. Where groundwater systems dominate, droughts ending in March have also been used that identify events that have dry winters. The longest historical drought events that cause system failure are a maximum of three years long, whereas the shortest are around 6-12 months. Longer droughts are documented in the historical record, however Spraggs et al., (2015) found that for the Anglian region, the 1890-1910 drought period was no worse than shorter, more recent droughts with respect to reservoir responses. Notwithstanding this, the 1890-1910 drought period highlighted the significant impacts of consecutive dry winters on water supplies through the unprecedented duration of low groundwater levels (Marsh et al., 2007).

The timing of termination, as well as rainfall events during, historical droughts, are critical in determining the characteristics of a drought and therefore how the system will react. For example, the 1976 drought ended in September and October with record rainfall (Alexander and Jones, 2001), whilst the 2010-2012 drought ended in April 2012, again with record rainfall.

Historical drought events of durations exceeding one year and up to three years have tended to have similar spatial coherence (Water UK, 2016). These longer drought events that span winters tend to be driven by large scale circulation patterns such as the North Atlantic Oscillation (NAO) and are generally more spatially coherent (Rahiz and New, 2012). In general, the Pennines act to reduce drought severity when droughts are focussed in the north of England and serve to prevent the spread of droughts from Yorkshire to Lancashire. Droughts focussed in Yorkshire tend to be coherent with eastern and south eastern areas, whereas droughts centred in the south east of England have a wider range of coherence across most of England (Water UK, 2016). There is a known west / north west and east / south east rainfall divide across the UK that manifests within drought periods as well as during average conditions. A number of shorter duration droughts (e.g. 1959, 1984) have been more locally focussed (Water UK, 2016).

2.6. Drought datasets

A summary of the different historical drought eras are presented below, based on uses within water resource applications:

- 1921 – present day. Drought events from 1921 onwards have been used as benchmark events or design droughts in water resource management planning in recent times (e.g. Marsh et al., (2007), Water UK (2016)). Whilst instrumental records do exist further back in time, the severity of drought events from 1921 to the present day and the spatial availability of instrumental data have meant that they have not been used extensively to test water supply systems within WRMPs and Drought Plans. There was a significant increase in the density of the rain gauge network in 1961, which has a dramatic impact on the estimates of rainfall within some catchments (Keller et al., 2015).

- 1760s – 1920. There are some instrumental records that cover this period (including gridded Met Office datasets going back to 1836 (Met Office et al., 2018)), however they are sparse and may not always be of good quality (Todd, 2014). There are issues related to a lack of instrument and observation standards as well as snowfall under-catch prior to the 1860s (Murphy et al., 2020b). Data may be partially reconstructed to improve spatial coverage (Spraggs et al., 2015) and evidence of drought events in this period may be supplemented with written documentation (Murphy et al., 2020a) or proxy records (e.g. grain harvests) that may not give a complete picture of the spatial extent, severity, duration or timing of the drought (Pribyl et al., 2012).

- Pre-1760s. Reliable instrumental records are sparse and may exhibit biases (Murphy et al., 2020b). Drought events are reconstructed through the use of proxies (e.g. tree-rings (Cooper et al., 2013), tree-ring isotopes (Loader et al., 2020)) or written documentation (Wetter et al., 2014). It can be difficult to understand the spatial extent, severity, duration or timing of a drought, particularly as reconstructions do not provide accurate information on extreme lack of rainfall and high temperature (Woodhouse et al., 2016), as well as not for the full calendar year (Wilson et al., 2013) and generally not covering winter climatology, which makes application to water supply assessments difficult.

The nature of water resource modelling in the UK is that high resolution weather variables are fed into hydrological and hydrogeological models and subsequently into water resource system models. A wealth of documentary evidence on historical droughts exists, however a means of reliably translating this documentary evidence into decision relevant datasets is not yet available.

Despite the requirement for high resolution weather variables, the recent enhanced Future Flows and Groundwater Levels project (eFLaG - Hannaford et al., (2022)) demonstrated that the data used as inputs into hydrological, hydrogeological and water resource models can vary from one water company to another (e.g. catchment average compared with gridded data). Some companies also use their own datasets, compared with publicly available datasets. This is a particular issue for computed datasets such as potential evapotranspiration and could have impacts on water supply systems in catchments with large baseflow components (Counsell, 2018). Historically, the water resource management planning process has not included a requirement to understand the impacts of dataset uncertainty beyond potentially being wrapped up in the supply-side contribution to Target Headroom. Some way of mitigating this source of uncertainty should be explored, either through adoption of a single dataset, such as that proposed within (UKWIR, 2017), or a planning approach that permits multiple lines of evidence. In any case, there should also be transparency and traceability about what and how data were used.

What we know

- **Droughts terminating in September or October tend to expose system vulnerabilities the most.**
- **The majority of English water supply systems can be classified as multi-season vulnerable, however their response to rainfall events can be complex due to system connectivity, conjunctive groundwater and surface water use and schemes that alter system resilience.**
- **Most water companies' test their systems against historical droughts that have occurred since the start of the 19th century. Recent droughts are more severe than those experienced prior to 1900.**
- **Of those events since the start of the 19th century, most of the worst historical observed droughts occur after 1920 and are generally three years or less in duration.**
- **Water supply systems evolve over time, often in response to the droughts they experience.**
- **Responses to drought are varied from one event to another and are a function of social, political, economic and physical factors.**
- Other users outside public water supply respond to drought events, potentially altering catchment resilience where interventions are of sufficient scale.

What we don't know

- **There isn't a full picture of changing resilience over time that covers all sectors, including from an environmental perspective.**
- **Historical interactions between sectors and public water supply system.**
- **The maximum severity or length of historical drought events.**
- **The occurrence of successive dry winters prior to ~1700s.**
- **An understanding of the spatial and temporal aspects of historical drought periods prior to 1890 at the catchment scale.**
- Likelihood and severity of droughts with successive dry winters, as well as underlying physical mechanisms for their propagation.

What we should know

- **Historical data related to other sectors water usage across different drought periods.**
- **Improved understanding of historical customer responses to drought and drought measures, focussing on quantifying the impacts of different drivers.**
- **Methods that can reliably extend the historical record (such as through improved palaeoclimatological techniques) to include events from the more distant past would be extremely valuable.**
- An agreed approach to dealing with observational dataset uncertainty, either through a choice of appropriate method, or a single agreed dataset.

3. Causes and variability of water supply drought in the UK

When considering the variability of water supply droughts, there are two main causes – natural variability and non-stationarity. Natural variability can be defined as the random or stochastic nature of events and results in differences between drought events. Non-stationarity can be defined as changes in the mean or variance of a time series, and results in changes in the causes of water supply drought over time (see Slater et al., (2021) for a detailed review of non-stationarity). Natural variability related to water supply droughts includes the stochastic nature of meteorological drivers and weather pattern persistence, that may cause droughts, as well as weather patterns that might drive the daily variance of water demand. An important omission in this section is the non-stationarity associated with future climate change, which is discussed in later sections. We know that there are numerous factors that contribute to water supply droughts in the

UK. These could be related to the supply, demand or vulnerabilities within the supply network itself.

3.1. Supply

These can include large scale meteorological drivers that may impact the weather, including the North Atlantic Oscillation (NAO), East Atlantic pattern (West et al., 2022), El Nino Southern Oscillation (Folland et al., 2015) and Sea Surface Temperatures (Serinaldi and Kilsby, 2012). These drivers can impact different geographical areas of the UK at different times of the year, depending on the state of the driver. For example, La Nina episodes may impact the severity of multi-year droughts in south-east England (Folland et al., 2015), while a positive phase NAO is associated with increased likelihood of summer drought (Folland et al., 2009). These drivers may also interact to enhance or moderate drought likelihoods, such as the interaction with the NAO and East Atlantic pattern (West et al., 2022). These larger drivers impact precipitation, soil moisture, potential evapotranspiration and open water evaporation via processes such as the persistence of weather patterns or position of the jet stream (Folland et al., 2009), which subsequently have impacts on hydrological (Svensson and Hannaford, 2019; West et al., 2022) and hydrogeological (Rust et al., 2019) responses, which can then impact water supply systems. As stated in the section above, these impacts may vary depending on the vulnerability of the water supply system and the nature of the weather variables. Despite the depth of research into the causes of drought in the UK, the relationships between large scale meteorological drivers and weather variables over England are still uncertain, largely due to the multiple drivers responsible for drought propagation (Folland et al., 2015) as well as the limited length of the data record (Slater et al., 2021). There is ongoing research in this area that is shedding new light on the combinations of multiple drivers that are likely to result in drought conditions (West et al., 2022). The topic of NAO and its influence on dry spells and drought is discussed in more detail within (Shaffrey, 2023).

As of WRMP24, there is a requirement to demonstrate system resilience to a 1-in-500 year return period drought (Environment Agency, 2021a) – usually droughts that are more extreme than those experienced historically. Droughts are a multivariate problem in that their frequency is a product of the severity, duration, timing and spatial extent. Consequently, the assessment of return period has moved away from estimation using input variables (such as rainfall) to system response (failure frequency). Such failure frequency is often termed the Level of Service (LoS), which is the frequency with which customers might expect to experience drought measures. Stochastic modelling approaches have been developed (e.g. Wilks and Wilby, 1999; Serinaldi and Kilsby, 2012; Dawkins et al., 2022) in an attempt to sample a greater range of droughts than those experienced historically and these have been applied extensively within UK water resource management. These modelling approaches differ from one framework to another due to underlying assumptions (such as the parameterisation of drought spell-length) that result in different drought characteristics (Chun et al., 2013; Dawkins et al., 2022). The use of return periods typically require the use of Extreme Value Analysis (EVA). There are a number of different methods of implementing EVA (frequentist versus numerous statistical approaches), which can have a material impact on the severity of the drought selected for a given return period and the treatment of uncertainty (Bristol Water, 2022). An example of this impact is shown in Figure 0.4. There are also questions around the assumption that all drought events belong to the same population and therefore can all be treated equally within a frequency analysis. The data that the stochastic generator are trained on can also skew the characteristics (frequency, timing, spatial extent, severity and duration) of droughts produced. Ensemble members produced by stochastic generators are usually treated as independent, however when dependence within drivers across ensemble members is considered, the sample size reduces, which in turn reduces the largest return period that can be reliably derived from the dataset. This is particularly relevant when trying to generate data at a national scale, where relationships between drivers and weather variables vary across the country, meaning droughts are not spatially coherent between regions. LoS can therefore vary greatly depending on the method selected as well as the events within the dataset.

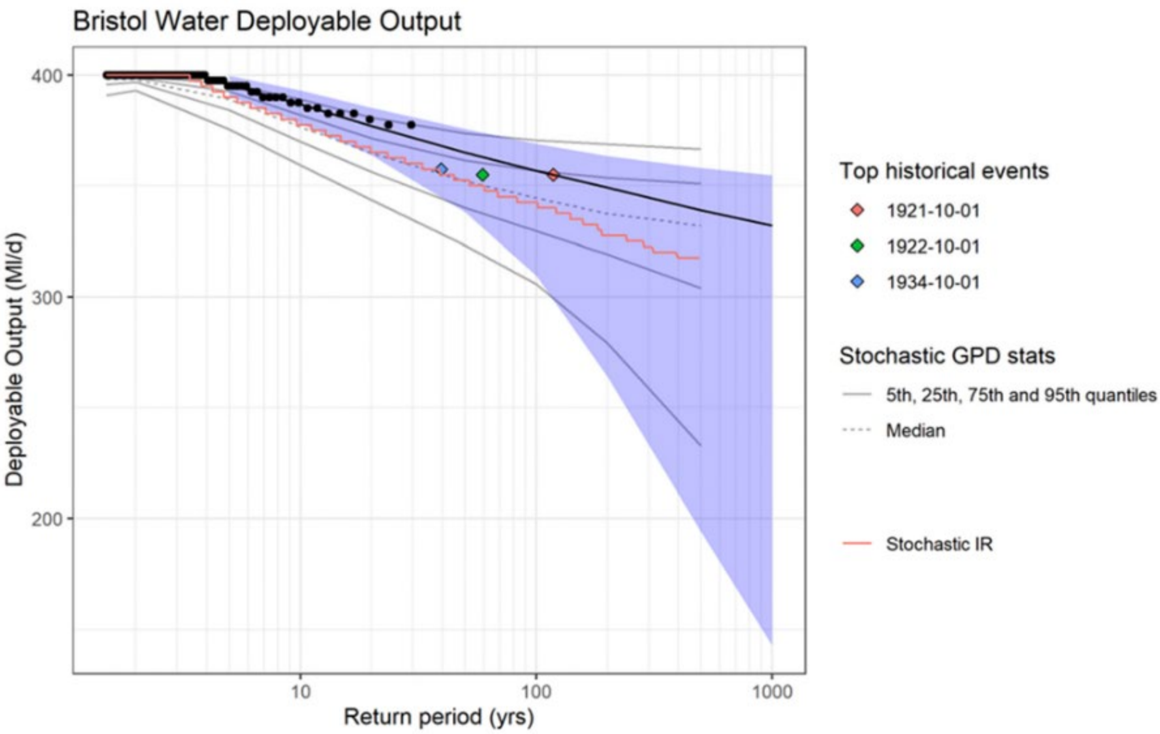


Figure 0.4: Comparison of EVA and inverse ranking approaches for the Bristol Water supply system. Historical inverse ranking is shown as points, historical EVA at the 95% confidence interval as the purple shaded area, stochastic inverse ranking with 400 replicates combined into one series as the red line, and stochastic EVA undertaken on each replicate as the grey lines. Source:(Bristol Water, 2022)

As well as stochastic approaches, synthetic events have been produced that are used to test water supply system resilience. These could include a third dry winter scenario (Spraggs et al., 2015; South East Water, 2019) or storyline approaches where historical events are altered, using physically plausible climate drivers (Chan et al., 2022), or through extension of historical events using other historical data (South West and Bournemouth Water, 2019) or developing a library of synthetic design droughts to systematically stress test a system (Environment Agency, 2015). These approaches tend to be more targeted towards exploring system vulnerability, as opposed to the large scale stochastic drought datasets that require a large number of model runs. There is difference in the way that these approaches tend to be interpreted by a modeller, in that large datasets rely on the use of metrics to understand the system, whereas smaller datasets permit the exploration of the individual events themselves. It is likely that the smaller datasets result in improved interrogation of system vulnerability.

Given the relatively short instrumental record, it is clear that historically documented droughts in the UK do not describe the full range of events that may be experienced. There are known differences between the stochastic frameworks used to generate alternative historical drought events that result in different drought characteristics (Chun et al., 2013). It is therefore likely that the full range of uncertainty in drought variability is not fully known, as it is dependent upon the assumptions within each statistical approach, as well as the uncertainty in the underlying input data. Whilst some stochastic weather generators are capable of producing events longer and drier than those in the instrumental record, there is a large amount of uncertainty in the characteristics of these types of events, particularly as they are conditional on the statistical methods employed within the generator (Brunner et al., 2021). Due to the lack of long, dry events within the instrumental record and the uncertainty in the statistical methods, the limits of plausibility of long and dry events are contested between different evidence sources (Wade et al., 2015). Whilst storyline approaches that provide counterfactuals of historical events are very useful in exploring events more severe than the historical record, they lack a systematic approach to exploring the causal underlying climate drivers and how these might change from one drought to another. Whilst water companies can build up drought libraries using these approaches to test their systems, it is likely that these libraries do not explore the full range of drought causes, variability and uncertainty. Instead, they more likely represent a set of outcomes that are conditional on the input data assumptions.

As well as the weather and climate, land use changes and management practices may exacerbate or mitigate the impacts of drought. For example, changes in forestry cover and growth over time have been shown to significantly impact runoff relationships (Birkinshaw et al., 2014; Afzal et al., 2021), agricultural crop choice may impact soil moisture deficits and alter river flow regimes (Afzal and Ragab, 2019) and increasing urbanisation may result in increased runoff response and reduced recharge (Afzal and Ragab, 2019), depending on interventions. Studies undertaken on land use changes are often tied to the specifics of the case study site (e.g. Birkinshaw et al., 2014) and therefore generalisation to other catchments, or even areas within the same catchment, can be difficult. In addition, when water resources and forest cover are viewed on a systems-level (particularly through the lens of forests influencing climate and weather), there is no consensus on the influence of forest cover on catchment water yields (Bennett and Barton, 2018). The assessment of the impacts of land use changes therefore have high uncertainty. In general, land use changes are not included within WRMP modelling. There may also be hazards related to land use, such as wildfire, that may cause variations in drought impacts through both space and time.

Climate change to the present day is also an important factor in more recent droughts, for example, modelled attribution work indicates the 2018 drought likelihood increased from less than 10% prior to 2000 to between 10% and 25% by 2018 (Lowe et al., 2018). This has implications for the types of techniques and approaches used to understand past, present and future risk from climate change.

3.2. Demand

3.2.1. Drought demand

As well as factors that influence the supply of water, the demand for water can also exacerbate drought impacts. Little is known about how domestic water demand responds under drought conditions (Manouseli et al., 2018), although insights may be garnered through tracking public internet search terms (Wilby et al., 2023). Water companies are continuously updating baseline demands via water resource management and drought plans as recent droughts expose new uncertainties (e.g. Affinity Water updating demand profiles due to the 2018 long summer drought exposing the vulnerability of high demand and low groundwater levels (Affinity Water, 2022) and South West Water experiencing unprecedented water demand due to increased holiday-makers as a result of Covid-19 restrictions in 2021 (South West and Bournemouth Water, 2022)). The evidence on the effectiveness of demand restrictions during drought events is also mixed, with Environment Agency (2013) concluding a 1-2% reduction in demand due to hosepipe bans during the 2012 drought, while UKWIR (2013) suggests that outdoor use of water reduced by 22%. The effectiveness of hosepipe bans can also change between drought periods, with water use reduced by 5-9% in 2006 (UKWIR, 2007). These differences between drought periods may be due to climatological factors (Manouseli et al., 2018) political (such as increasing demand in Yorkshire as a result of perceived mismanagement (Bakker, 2017)) or socio-economic factors (Taylor et al., 2009) and may be linked to experiences of previous droughts, may be short lived, and may also extend beyond the period of implemented restrictions (Manouseli et al., 2018). However, in general, the evidence tends to suggest that households are more likely to reduce demand through behavioural changes and technological interventions rather than temporary use bans (Manouseli et al., 2018; Wilby et al., 2023). There are also documented gaps between attitudes and behaviour related to water demand that can have impacts on the success of information-based campaigns to reduce consumption (Manouseli et al., 2018). In relation to the socio-economic and political factors, despite advances in techniques such as agent-based modelling, there are still significant limitations in the prediction of impacts, including the large number of assumptions of behaviour without available data for calibration (e.g. Darbandsari et al., 2020).

As well as exacerbating drought impacts within a drought, the magnitude of non-drought water demand throughout the year may restrict the water available for use within a catchment or supply network, reducing the resilience of a system to deal with periods of dry weather. Examples include household behaviour, favoured types of energy use (Environment Agency, 2011) and agricultural demand.

3.2.2. Everyday demand

There has been a substantial amount of research undertaken into the factors that affect everyday (non-drought) domestic water demand. This research demonstrates that a large number of factors can influence domestic water demand, including climate (temperature, precipitation (Parker and Wilby, 2013), evapotranspiration (Goodchild, 2003)), socio-economic (Manouseli et al., 2018), day of the week (Kowalski and Marshallsay, 2005) and dwelling characteristics including garden size (Pullinger et al., 2013; Manouseli et al., 2017). These factors are often nuanced and their relationship with water demand can be complicated (e.g. household income (Manouseli et al., 2018)), making estimation of demand difficult (Parker and Wilby, 2013). These non-drought factors may have a larger impact during periods of below average rainfall that span multiple years, which may be exacerbated by public perceptions of drought caused by non-arid conditions (Marsh et al., 2007).

There are also non-stationarities that may alter impacts over time, introduced through interventions to reduce demand. It is known that metering can reduce demand by between 7 and 35% (Manouseli et al., 2018), and that behaviour change during drought periods (enforced by hosepipe bans) is more prevalent in metered households (UKWIR, 2013). Such interventions may therefore have potential consequences for both peak and annual demand constrained water supply systems. The installation of water efficient technologies into existing housing produces large reductions in demand, however these reductions vary substantially from one household to another (Manouseli et al., 2018). Whilst historical water demand patterns have been studied and some of the factors that determine this demand are known, a limited amount of work has been undertaken to apply these to models that can help understand what the range of historical uncertainty might be (Manouseli et al., 2018).

3.2.3. Agricultural demand

Agricultural demand varies depending on the type of farming and practices. Use for irrigation is approximately 1-2% of total abstraction in England and Wales, but is significant due to the timing of abstraction in drier months and geographical location

(Weatherhead and Howden, 2009). This significance is reflected in the public interest in irrigation through increased internet searches during drought events (Wilby et al., 2023). Water restrictions on growers can have severe financial implications (Knox et al., 2000) and creates the need for headroom within abstractions (Sutcliffe et al., 2021) that can alter wider system resilience. Spray irrigation in these areas is also generally 100% consumptive.

Water trading, being trialled as part of abstraction reform, can also result in changes in abstraction behaviours (DEFRA, 2019). Food sovereignty may also drive changes in policy that impact water use (Weatherhead and Howden, 2009). There is little information available on the temporal patterns of the use of different water sources, and how growers switching from mains to directly abstracted water may impact resilience during droughts (Knox et al., 2020). Drought vulnerabilities within an area (such as the East of England) can also vary across sectors such as agriculture (Environment Agency, 2020), highlighting the need to create drought datasets that reflect a multisectoral perspective.

3.2.4. Industrial and energy demand

Industrial and energy demands have varied significantly throughout the UK, and will likely do so in the future. The decline of industry since the 1970s and associated reduced demand have resulted in water availability in areas not previously projected to have a surplus (Archer, 2003).

The distribution of certain water-hungry uses such as chemical and heavy industry (among others) are not equally distributed throughout the UK (Environment Agency, 2020). The 1959 drought impacted chemical and heavy industries in the Tees Valley (Taylor et al., 2009), and during the 1976 drought, there was an increased importance in protecting industry from impacts (Grecksch and Stefán, 2018). In addition, the amount of water consumed depends on the industry and water use. For example, evaporative cooling for electrical generation is almost 100% consumptive, whereas hydro-electric, fish farming and through-flow for cooling return almost all water used, generally close to the abstraction point (Weatherhead and Howden, 2009).

3.2.5. Spatial aspects of demand

There is also an important spatial aspect to water demand in the UK, with water demands not equally distributed across any sector. This can have important implications during different types of droughts at different times. For example, the 2020 summer drought coincided with Covid-19 lockdowns, resulting in unusually high seasonal demands in areas such as the Thames Valley (Thames Water, 2022a) and the stagnation of industrial demand in the north east of England around the time of the construction of Kielder Reservoir which resulted in a large surplus in supply (Archer, 2003). The interaction of demand and the vulnerabilities of the supply network (which may be a result of geographical location, nature of supplies or network connectivity) creates a high degree of variability in the vulnerability of supply networks over time and space. This variability tends not to always be within the control of the water companies.

3.3. Supply network

There may also be causes that originate from within the water supply system itself that may exacerbate drought impacts. Water storage for supply or downstream compensation may have to be traded-off against hydropower generation or flood risk prevention at particular times of the year (Douglas, 1988). These trade-offs are likely to be implemented in an imperfect way, such as forecasting releases to make space for flood attenuation within a reservoir based on predicted frontal systems in the spring. If these forecasts are inaccurate, water supply storage may be sacrificed and exacerbate summer drought conditions (Environment Agency, 1995).

There may also be outage issues, which include water quality issues, limited capacity of the distribution network, leakage, maintenance of the network, and dam subsidence which can all contribute to drought impacts. As with demand and supply related issues, these can vary from one drought period to the next. These issues are often included within an outage assessment as part of a water company WRMP, however these are undertaken in an aggregated, rather than systems-level, analysis and so potentially miss combined risks despite the use of Monte Carlo simulations (UKWIR, 1995).

3.3.1. Leakage

National Infrastructure Commission (2018) highlights the aspiration to halve leakage by 2050, with the aim of increasing the volume of water available during drought events. Currently, approximately one fifth of treated water is lost to leakage (National Infrastructure Commission, 2018), with environmental implications including increased energy cost per volume of water supplied and implications on supply for households in rural areas (UKWIR, 1997). Such interventions are sometimes termed low or no-regret (Birks et al., 2023) and whilst leakage from mains water is significantly important in aquifer recharge, particularly in urban areas (Yang et al., 1999) that may support river flows with water of a high quality, depending on the specifics of leakage locations and catchment characteristics, there is little analysis available on a systems level to determine the costs and benefits of leakage reduction where this is the case.

Leakage often increases during drought events due to ground shrinkage (Durant, 2015). Drought triggered leakage events are spatially unequal, tending to occur where clay soils dominate and often where drought conditions are most severe, exacerbating drought conditions (Durant, 2015). Consequently, measures are often taken within a drought to manage leakage events. As well as impacts within and outside drought events, there may be general changes over time. These may be technology driven (improvements in tracking and tracing leaks), policy driven (e.g. the halving of leakage by 2050 driven by discourse particularly since the 1995 drought (Miller and Yates, 2006)), related to the age of infrastructure (UKWIR, 1997) or the method of calculation as to the sustainable economic level of leakage (Ashton and Hope, 2001).

What we know

- There are lots of different and overlapping climatological drivers of water supply drought.
- Land use changes can have a large impact on supply, however extrapolation outside study catchments is difficult and largely not included within WRMP modelling.
- There is a large amount of uncertainty with stochastic approaches, including in the methods used to apply them to water resource planning, including EVA.
- Recent droughts are improving understanding of present day demand patterns.
- There is a large uncertainty in the historical impacts of demand reductions.
- A large number of factors can influence household demand. These have altered over time and change system resilience for annual-demand critical systems.
- The distribution of water-hungry uses is unequal around the UK, with implications for drought planning (protection of industries from drought impacts), as well as long term planning (when industries collapse).
- The spatial aspects of demand during a drought can have a large impact on drought resilience and can vary greatly from one event to another.
- Factors within the supply network itself can exacerbate drought conditions, such as leakage, outage, water trade-offs and forecasting.

What we don't know

- A full understanding of climate drivers for all historical events that can explain all variability.
- Understanding of land use changes and related hazards (such as wildfires) on water resource modelling.
- Uncertainty in stochastic approaches is not fully explored.
- The full envelope of plausible drought events that could occur in the UK.
- The full envelope of plausible peak demand. The uncertainties associated with demand are complex and not fully understood.
- The interaction of risks related to outage and leakage on a systems-level.

What we should know

- Systems-level understanding of land use changes on water resource modelling to understand the magnitude of the uncertainty.
- The different stochastic approaches that exist and their suitability to UK water resource planning problems.
- Understand how demand may vary as a result of behavioural response during and following different drought events.
- Whether the interaction of outage risks in a deployable output, aggregated approach is similar to that when assessed on a systems-level.

4. Future water supply droughts

This section examines the characteristics of future water supply droughts in terms of climatology (affecting water supply), demand and pressures on the network itself. Future climatology is assessed through the outputs of current climate models, the processes within the models and a review of approaches for incorporating climate change information into assessments.

When referencing future scenarios, it is useful to introduce uncertainty. Dessai and Hulme (2004a) outline three different types of uncertainty – epistemic (that which can be reduced through improved knowledge), aleatory (irreducible natural, or stochastic, variability) and reflexive (iterative human behaviour). Deep uncertainty is a term that encompasses all of these types of uncertainty, and is a condition where parties do not know, or cannot agree on models that link inputs to outputs, the probability used to represent uncertainty or the value-sets to attach to outcomes (Lempert et al., 2003). Such a condition can remove the requirement for probabilities or likelihoods attached to future scenarios (Dessai and Hulme, 2004). Indeed, where probabilities are used, they are conditional on the climate models used in the analysis. There is therefore a lot that we do not know about how a future will play out and it is high likely that any projected future world states will not materialise as they are envisaged in scenarios.

4.1. Future climatology - outputs

The main messaging from the UK Climate Projections 2018 (UKCP18) is that of warmer, wetter winters and hotter, drier summers (Lowe et al., 2018). This messaging is taken from the changes between the baseline period and future periods and describes the general, likely direction of travel under different emissions scenarios. This is also the main messaging when mean monthly changes are computed between baseline and future periods for input into hydrological, hydrogeological and water supply modelling. This messaging is consistent with UKCP09 and UKCIP02 (Lowe et al., 2018). The UKCP18 projections indicate that the uncertainty within an ensemble projection is greater than the uncertainty between emissions scenarios, and the uncertainty between different UKCP18 products (UKWIR, 2021). Whilst certain aspects of climate modelled processes have been improved from previous generations to those used in UKCP18 (Lowe et al., 2018), the uncertainty within UKCP18 has not been reduced. In actuality, the range of precipitation values in the probabilistic projections in UKCP18 is greater than in UKCP09 (Kay et al., 2020). The timing of future changes are also uncertain, and depend on the climate model and greenhouse gas concentration used within scenarios.

Current evidence points to UK wide droughts being possible or more likely (Wade et al., 2015; Murgatroyd and Hall, 2020; Reyniers et al., 2022) as well as droughts with a large spatial extent becoming more severe (Rudd et al., 2019). There is also evidence that extreme droughts and associated reservoir levels will worsen in 80 studied catchments across England and Wales (Dobson et al., 2020). Drought durations may increase in the future, although there is an increased likelihood of shorter duration events (Rahiz and New, 2013; Reyniers et al., 2022). Drought frequency is also expected to increased, particularly in England and Wales (Reyniers et al., 2022).

Evidence relating to the future occurrence of dry winters is uncertain, with a likely reduction in occurrence (Wade et al., 2015; Lowe et al., 2018). Notwithstanding this, there is still a chance of dry winters, as well as a chance that they occur successively. There has been little work undertaken on the likelihood of future dry winters on the basis that they tend to reduce in frequency in the future under most emissions scenarios. Work by Mansour and Hughes (2018) on the effects of climate change on groundwater recharge in the UK identified a shortening of the recharge season, potentially increasing drought vulnerability, as well as increasing the possibility of groundwater drought should rain fall in a month where recharge is subsequently more reliant. The use of change factors also hides the inter-annual variability within the climate outputs, and therefore can only be ascertained when looking at climate projections that are temporally coherent. Moving beyond change factor and mean change type approaches, work has been done looking within the UKCP18 projections themselves at the evolution of drought, indicating that there is an increase in meteorological drought severity for durations of 3 to 36 months (Hanlon et al., 2021).

We know that bias correction can, and often is, implemented before climate model outputs are used in impact and vulnerability assessments, that different bias correction methods (Gohar et al., 2017) and baseline periods (Lafon et al., 2013) can give different outputs, and that bias correction can alter drought characteristics such as drought duration (Maraun et al., 2021). We also know that variable-based bias correction makes assumptions, including that the relationship between baseline and future periods is time-invariant (Maraun et al., 2017). The issues with bias correction of variables have led to calls for process-based bias correction (Maraun et al., 2017), as well as identifying the biases propagated through downscaling using regional climate models (Addor et al., 2016). Differences between bias correction methods can be larger than the differences in GCM to RCM downscaling (Laux et al., 2021). A large number of bias correction and downscaling techniques and tools exist and have been critiqued over the years (Maraun, 2016), often with the conclusion that it is fraught with difficulty and implementation should not be undertaken without careful consideration of the biases within the climate model and the end use (Maraun et al., 2017). Indeed, there is an argument that climate models should compare reasonably well to observations prior to bias correction (Kundzewicz and Stakhiv, 2010). This topic of the removal of models, including those with large biases, is discussed in more detail below. More work is therefore required to understand the specific requirements for UK water resource planning to ensure any move away from the change factor approach represents an improvement in capturing the processes and variables of interest.

Change factors are a very simple form of bias correction and downscaling, and assume that the climate models more accurately simulate relative changes than absolute values, and that these biases are constant through time (Fowler et al., 2007), regardless of how biased the model may be during the baseline period. Change factors also only alter the mean, maximum and minimum values of weather variables, and assume that the spatial extent, variability and frequency of events are the same as those in the dataset that is being perturbed (Diaz-Nieto and Wilby, 2005; Fowler et al., 2007). This process does not therefore accurately reflect characteristics from climate models such as dry winters. This can have impacts on the complexity and therefore diversity of drought events that are produced as scenarios for input into a water resources model. We know that older CMIP5 models struggle with drought persistence when examining variables within outputs (Maraun et al., 2017; Moon et al., 2018). It is unclear if this has been improved within newer CMIP6 models with evident implications for water resource management if moving beyond change factor approaches.

Current planning guidance for the inclusion of climate change within water resource assessments in England and Wales recommend the use of mean change factors. These monthly change factors are then applied to severe, 1-in-500 return period (determined from system impacts) drought events to provide a 1-in-500 plus climate change drought event. Given the post-processing nature of applying these change factors, this method is not dynamic and it is unclear how plausible the resulting drought might be. There are also limited ways of readily assessing the plausibility of future events. The 1-in-500 plus climate change drought is therefore probably an extreme scenario for summers, given the relatively short dry events experienced historically, the perturbation of these through stochastic generation and the general messaging from change factors of hotter, drier summers. However, the approach is not necessarily one that describes what a future drought may actually look like. No work has been done to contextualise these future drought events with those available through other lines of evidence such as bias-corrected climate model outputs.

4.2. Future climatology – processes

In the development of climate products for the UK, UKCP model evaluation is undertaken during the model build (Lowe et al., 2018), but not necessarily for all variables of interest or at a temporal scale or timing that is useful for water resources. CMIP5 Global Circulation Models selected as part of UKCP18 that were not developed by the Met Office were screened in a qualitative and quantitative way for both global and regional performance (including biases in sea surface temperatures or south-westerly flow over the UK), as well as to ensure a diversity of model structural uncertainty (Lowe et al., 2018). Such an assessment represents an evaluation of general suitability to applications within the UK, however not necessarily directly relevant to UK water resources. Detailed information on CMIP5 model selection as part of this process is not publicly available. This represents a different process to that undertaken in other countries where model evaluation is undertaken on phenomena of interest (e.g. drought, tropical cyclones, surface winds etc.) such as those undertaken in Australia (CSIRO, 2015; Moise et al., 2015). Much of this sector evaluation has been undertaken in the UK since the publication of UKCP18 (e.g. Pope et al. (2021); Cotterill et al. (2022)). The use of climate processes and teleconnections is relatively new and there is evidently a large amount of work to be done to understand how to relate these to water resource planning approaches, both in terms of future changes, using the processes to understand plausibility of future changes and how to implement this within planning approaches. The latter is discussed in the last chapter.

Historically, climate model evaluation for a particular purpose has focussed on the comparison of historical observed variables with those from a climate model. There are a number of issues related to this practice, particularly in relation to droughts, including comparisons of data at different scales (Wilby, 2010), as well as the problems with short observed records that may misrepresent natural variability and allow the drawing of spurious conclusions (Armstrong et al., 2020). Consequently, more recently there has been a focus on a comparison of the processes, rather than the variables themselves. For example, Cotterill et al. (2022) demonstrate that there is a trend for an extension of summer into autumn in the future based on the frequency of weather patterns within the latest CMIP6 and UKCP18 GCM PPE members. Such an analysis may have important implications for water resource planning due to drought termination timing which, along with a link to the underlying weather patterns, permit an (albeit limited) assessment of the plausibility of such changes. Pope et al. (2021) indicate that weather type persistence within the UKCP18 GCMs correspond well with observed frequencies, and that in the future under an RCP8.5 greenhouse gas concentration scenario, weather types could shift to more settled, blocking patterns in summer, and more cyclonic, westerly weather types in winter, consistent with the UKCP18 headline findings on weather variables and with data outside UKCP18 (De Luca et al., 2019). Whilst we have an understanding of how existing weather types may change in the future, little work has been done to understand whether new types will emerge and what a change in the definition of these patterns might mean (Pope et al., 2021). Consequently, as well as static methods of analysis, dynamic methods are required to deal with non-stationarities in the underlying drivers of change. There is a lack of interrogation of the wider process such as the North Atlantic Oscillation within climate models that are known to cause variability in UK droughts in specific relation to water supply system vulnerability. There is also almost no understanding of whether drivers of drought may change in the future. Work is, however, currently being done to understand some of these wider processes in relation to meteorological indices (Barnes and Brierley, 2022).

The removal of models due to poor performance is contested. There is a school of thought that the removal of models from an ensemble due to poor relative performance results in a biased selection of models that means probabilities can't be assigned and so the full range of uncertainty should be considered (Guerreiro et al., 2017). There may also be models that share common processes and code that mean they are not truly independent (Knutti et al., 2013). In addition, the multi-model mean tends to perform well (Bishop and Abramowitz, 2013). There is a counter argument that models that do not adequately model the area, variables or processes of interest should be removed, or weighted on the basis of credibility. These assessments could include quantitative or qualitative considerations. There are potential planning implications that result from the removal or inclusion of models based on plausibility (Zhang et al., 2021). Whether models are removed or not, there is a significant knowledge gap in understanding whether models that skilfully model the past also skilfully model the future, which is currently assumed in many climate model evaluations (Moise et al., 2015).

4.3. Future climatology – alternative methods

A number of existing and emerging methods are being developed that use climate model information and / or historical data to elicit scenarios. Storyline approaches have been proposed as a means of rooting climate impacts in physical plausibility, bounding uncertainty, and resulting in decisions that are potentially more rooted to physical aspects of climate change (Shepherd et al., 2018). Climate analogues, whereby historical climate information from one location is mapped to another that is expected to experience similar conditions in the future, have been demonstrated as a means of generating scenarios in a heuristic and easy to explain way (Hallegatte et al., 2007), however there are some issues with using this approach, mainly due to potential differences in the driving physical processes between locations. Stochastic approaches have been applied to climate model outputs in order to generate variability around time slices of future climate (e.g. Glenis et al. (2015)). There are also plans to incorporate the UKCP projections into the Advanced Meteorological Explorer to generate such future scenarios for global mean warming levels (Dawkins et al., 2022). The Unprecedented Simulated Extremes using Ensembles (UNSEEN) approach has been applied to extreme rainfall events (Thompson et al., 2017; Kelder et al., 2020), permitting increased understanding and contextualisation of historical events. More recently, this has also been applied to the risk of drought in the UK, along with a link to storyline events by extending similar events to those in historical sequence with plausible future events generated through the UNSEEN approach (Chan et al., 2022). Wade et al. (2015) examined the limits of plausibility for droughts through the H++ method, attempting to derive maximum changes in terms of severity and duration for the UK through the assimilation of multiple lines of evidence, including paleoclimate, climate change models and historical information, however climate modelling has evolved since this study was carried out. Notwithstanding these studies, there remains a lack of understanding around the plausibility of drought extremes both spatially and temporally, as well as water resource relevant drought characteristics such as termination. Aside from storyline-type counterfactual approaches (albeit without strong links in physical plausibility), none of these methods have been used in WRMP or drought planning in the UK.

4.4. Future demand

Based on general information within UKCP18, there may be a movement towards peak demand issues through increased aridity and high temperatures. The probability of experiencing a summer as hot as 2018 could potentially double by 2050 depending on the emissions scenario (Lowe et al., 2018), resulting in more frequent implementation of drought measures if no adaptation is undertaken.

We know that there is a large uncertainty in the range of future demand forecasts, in a similar way there is a large uncertainty in future water supply projections (up to 50% across UK water companies (Manouseli et al., 2018)). As an example, the Thames Water draft WRMP24 household demand forecast for the Thames Valley contains approximately 25% uncertainty by the year 2045 at the 95% confidence interval based solely on bootstrapping parameter values within a single model, based on a best estimate of future trends (Thames Water, 2022b). There are also large spatial variations in future demand uncertainty (Manouseli et al., 2018). There have also been few attempts at integrating climate change uncertainty related to demand management into water resource planning (Anderson et al., 2018). Non-household demand estimates for regional groups, as well as a discussion of some of the key factors in uncertainty and future changes for different sectors can be found in Environment Agency (2020).

There are also large uncertainties related to changes in future peak and annual demands, related to the uptake of water efficiency measures. There is little evidence to understand how changing demand annual demand patterns might impact system resilience during drought events, as households are less able, or less willing, to make as large a reduction in consumption when demand is already low, although there are some lessons from other countries where highly efficient, optimised systems can result in decreased resilience (Rodina, 2019). This is a critical aspect of system modelling that has not been considered within English WRMP planning to date due to the frameworks used.

4.5. Future system pressures

Where catchment scale studies on land use and climate change have been undertaken, these have shown varying responses, with climate change impacts exceeding land use change impacts (Afzal and Ragab, 2020). Future droughts may also be exacerbated by decreasing water quality as a result of less dilution in summer and increased sediment concentrations in winter due to increased storm frequency and severity (Weatherhead and Howden, 2009).

It is acknowledged that the major drivers for future change are likely to result from population, socio-economic, climate and technological changes (Weatherhead and Howden, 2009). The uncertainty in these drivers is large, however through the increased use of scenario development, linkages between drivers, impacts and responses are reasonably well understood where linkages are simple. For example, a move towards hydrogen energy may drive increased water demand in the order of 15 – 20% (Water UK, 2022), although this will depend on the dominance of green, blue or grey hydrogen and the embodied water from energy production for electrolysis (Beswick et al., 2021). However, where feedbacks exist, there is less understanding of how future states might emerge. This has led to the increased call for digital twins that could model these feedbacks to improve understanding (Bauer et al., 2021).

A limited amount of research has been undertaken on land use and future climate change scenarios. Where studies have been undertaken, these are often on a catchment scale (Afzal and Ragab, 2020). There are known feedbacks at a systems level between climate change and land use, for example, changes in stomatal resistance due to widespread cropping and farming practice changes based on current projections (Ritchie et al., 2019). There are also large woodland creation targets for England that will likely result in areas of agricultural land being afforested (UK Government, 2021). This will have impacts on runoff regimes and are a direct result of human responses to present and future threats from climate change and biodiversity loss.

What we know

- **Projections generally indicate a likely increased (decreased) vulnerability for single season (multi-season) systems using mean change factor approaches.**
- **There will likely be increased spatial coherence, frequency and severity of events, based on evidence from available climate scenarios. Drought duration may increase, however there is a greater likelihood of increased shorter duration events.**
- **The 1-in-500 year return period drought plus climate change likely misses the changes in variability, timing, duration and extremes of droughts within climate models. We also know that climate models have limitations as to**

how they simulate drought events. It is impossible to assign a meaningful return period on such events, or even understand if they are plausible.

- **Dry winters are still possible in the future, and the lengthening of summer and shortening of winter means some aspects of drought vulnerability are being missed when using change factor approaches.**
- **There are lots of different bias correction and downscaling approaches. Different approaches have different impacts on corrected drought characteristics.**
- Peak demand may increase as a result of climate change increasing aridity and temperature.

What we don't know

- **The likelihood of dry, or successive dry winters, in the future.**
- **We don't know the impacts of moving away from a change factor approach. Any newly adopted approach should represent an improvement in those characteristics that are important for UK water resource systems.**
- **The limits of plausibility of existing climate change projections for water resource approaches.**
- **The planning implications of removing implausible climate models from assessments in the UK.**
- **There are large uncertainties associated with future demand, both in terms of household and non-household demand.**
- **A number of methods exist to incorporate climate change information into water resources planning, however these are yet to be tested within a WRMP framework.**
- **System resilience issues arise in efficient, optimised systems. It is unclear how resilience to drought events would be affected by demand side efficiencies.**
- There are numerous future pressures originating from within the wider catchment that may impact the availability of water, both within drought and normal conditions. The uncertainties associated with these are large and generally only modelled where specifically known to be an issue.

What we should know

- **Greater understanding of the extremes within climate projections, linked to the plausibility of these changes in terms of processes, rather than variables.**
- **The planning implications of removing implausible climate models from assessments in the UK.**

- **Interactions between high water efficiency, customer willingness to reduce demand during droughts and system resilience.**
- **A greater understanding of the impact of catchment scale pressures on public water supply.**
- Understand the specific requirements for UK water resource planning to ensure any move away from the change factor approach represents an improvement in capturing the processes and variables of interest.

5. Water supply drought impacts – present and future

A large amount of work has been undertaken at different scales across England into the impacts of climate change on water supply. These analyses have generally been performed at national, regional, water company, supply system scales. This section reviews impacts at these scales and summarises the approaches used. The understanding of impacts is linked very closely to our knowledge of future and historical droughts. The knowledge base highlighted as part of the historical and future sections of this review therefore lead on to the conclusions drawn as part of this section.

5.1. National level

Current water supply is mostly resilient to the worst historic drought, however in order to maintain resilience at this level to 2050, taking into account increases in population and environment and climate pressures, an additional 2,700-3,000 megalitres per day (Ml/d) would be required in England (National Infrastructure Commission, 2018). Severe and extreme droughts would result in further water being required, the spatial distribution of which is mostly required in the south and east of England (National Infrastructure Commission, 2018). Despite this surplus, there is a high level of current risk across many regions in the south and east of England, largely due to the risk of a drought occurring over the next 25 years that is worse than those experienced historically (Water UK, 2016). Increasing drought resilience to a 1-in-500 year drought event is projected to result in a reduction in England's deployable output of around 1,140 Ml/d, over 2.5 times the current surplus (HR Wallingford, 2020). The general consensus is that a twin-track approach of reducing demand and increasing supply to areas in need is required (National Infrastructure Commission, 2018; HR Wallingford, 2020).

The National Infrastructure Commission (2018) report was published in order to draw attention to preparedness versus reactionary planning, however the report raises an important point about the disconnect between the modelling within long term planning and the reality of decision making within a drought. There is a need to better represent these nuances within assessments to fully understand the impacts of droughts, as well as the ability to recover from such events.

All of these national scale studies tend to follow a top-down approach which misses the opportunity to understand system vulnerability, particularly when considering severe long duration events at a national scale that may cause problems with temporary transfer of water between affected areas.

5.2. Regional level

There are large differences in the public water supply deficit between regional groups in England by 2050, with the greatest impacts in the south east and east, of which climate change only comprises 11% of the additional water requirement (Environment Agency, 2022). Other, larger contributors are reducing abstraction for the environment (policy decision), increased demand from population increase and increasing drought resistance (Environment Agency, 2022).

5.3. Water company level

There is a large range of uncertainty in present day impacts within water company WRMPs. For example, the stochastic ensemble uncertainty at the 1-in-500 year return period for Bristol Water represents approximately 40% of deployable output (Bristol Water, 2022). Or, conversely, the median DO at the 1-in-500 could plausibly be as low as a 1-in-20 year return period and as large as beyond a 1-in-100 year event. This is a similar number when the uncertainty in the historical record is considered. There is therefore a large uncertainty, particularly when using return periods to determine drought impacts.

Many groundwater sources have been designated as insensitive to climate change within water company WRMPs (HR Wallingford, 2020), based on the general messaging of warmer, wetter winters from the UKCP programme that indicates a reducing likelihood of dry winters. This is contradicted by evidence within (Mansour and Hughes, 2018) and therefore impacts of climate change of groundwater sources are likely currently underestimated. This also has implications for issues such as groundwater – surface water interaction, which has only recently been incorporated into relevant water company plans (Affinity Water, 2022).

There are significant spatial differences between different water companies (HR Wallingford, 2020). In addition, within water companies, the nature of supplies, their vulnerability, and the return of effluent to different catchments, or parts of the catchment results in different spatial manifestation of impacts (HR Wallingford, 2020).

5.4. System level

Environment Agency (2015) investigated the use of a generalised typology of droughts with which to test water supply system resilience. For those catchments tested single-season systems tended to be most sensitive to short duration droughts, and tended to fail reasonably quickly once a threshold was crossed, albeit once large rainfall deficits were experienced. Multi-season systems tended to fail slower, however experienced failure across a range of drought durations. Conjunctive use systems tended to be more resilient than those that relied on a single source or similar sources of water. There is a known systems-level issue of water quality and future climate change (Weatherhead and Howden, 2009; Hutchins et al., 2018). In long term planning, this is often lumped into an outage assessment, however this is often included within target headroom within a Deployable Output type approach, which ignores the resilience issues within a drought related to water quality. Given the likely exposure to increased water quality and other outage related incidents, an improved method for incorporation of these uncertainties into long term planning is required.

5.5. Approaches

All of these studies used mean change factor approaches to perturb events that could be expected to occur in the present day. The limitations of using mean change factors are discussed above. Some studies have used different approaches, however these have largely been within the academic literature, demonstrating the use of such datasets. Borgomeo et al. (2014) used a non-stationary stochastic approach (however still based on change factors) using the London supply system to arrive at similar conclusions of the need for a twin track approach, largely due to a 50% likelihood of exceeding planned Levels of Service by 2040. Roach (2016) resampled the Future Flows bias corrected climate projections to produce a large number of transient future projections within a scenario-generation step for decision making under deep uncertainty.

There are some commonalities across a number of these studies. They are all undertaken using large datasets and tend to aggregate impacts. Doing so tends to reduce scrutiny on individual events and risks missing system-relevant learning about vulnerability and resilience. It may also hide site specific issues related to biases and errors in datasets (Wilby et al., 2017). As with all approaches, they all have limitations, such as aggregation of outputs and uncertainty such as DO type methods, reduced spatial or temporal resolution or a small range of uncertainty in future climate or demand projections. The impacts are also scenario led, and so these results are conditional on the scenarios used. There is a lack methods to integrate these approaches to gain a full understanding of integrated risk.

Whilst future droughts will significantly impact many water supply systems, other factors, such as the water that is left for the environment after abstraction, may have a larger impact (HR Wallingford, 2020). Future water management and environmental flow policy are therefore critical in determining who or what bears the burden of reduced flows. It is clear that for many English catchments, trade-offs between stakeholders will be required (even where these trade-offs already exist), with possible solutions including smart licensing (Wilby et al., 2011).

What we know
• **Impact assessments are hampered by the upstream issues of bias correction and downscaling, hiding issues such as the sensitivity of groundwater to climate change.** • **There is large uncertainty in the present and future impacts due to a lack of knowledge of interacting risks, as well as how these risks may change into the future.** • **There are significant spatial differences in impacts across different scales.** • **Impact assessments tend to aggregate uncertainty to provide a "best-estimate" of impacts based on the available climate change projections.** • Future water management policies may have larger impacts than climate change.
What we don't know
• **Many impact assessments use large datasets and aggregate impacts. This may reduce scrutiny on individual events and system vulnerability as well as site specific biases and errors.** • **The specific water management policies of the future may be unknown.** • It can be difficult to combine outputs across different assessments to improve understanding due to differences in methodologies.
What we should know
• **Methods for incorporating large amounts of uncertainty into long term planning.** • There is a need to examine more diverse events in detail to understand system response.

6. Planning approaches – present and future

We know that the water resources problem is one that is difficult to solve due to a large number of competing interests, limited resource availability, increasing pressures (such as climate change, pollution, water reliability etc.) and the requirement to provide sustainable and cost-effective solutions that are equitable across generations. This has been termed a “wicked problem” by many (e.g. Ben-Haim, 2019; Siders and Pierce, 2021), meaning there is no right or wrong solution (Rittel and Webber, 1973). A large number of approaches have been sought to solve this problem. This section outlines the metrics and approaches currently available to assess water supply system performance, before reviewing the general frameworks of top-down and bottom-up approaches. Given the nature of public policy decision making, choosing an approach is also discussed, as well as possible future approaches.

6.1. Metrics and risk-based planning

Before introducing decision making methods, it is useful to first consider the metrics against which system performance is assessed. These have traditionally been considered to be reliability, resilience and vulnerability (Roach, 2016). Reliability is defined as how likely the system is to fail, resilience as the recovery from a failure (although resilience is a difficult term to define) and vulnerability as the consequences of failure (Roach, 2016). Current LoS methods can be considered to use reliability as the main performance metric, however vulnerability is intrinsically incorporated through the definition of a LoS as well as through the use of Drought Response Surfaces. Resilience has been used in conjunction with the 1-in-500 year return period (Environment Agency, 2021b), however the assessment itself does not differ from a LoS type approach.

Robustness is a planning approach whereby future options are tested against a number of scenarios and those options that perform best against a wide variety of scenarios are taken forwards (Lempert et al., 2003). Robustness is at the heart of a number of decision making methods, and tends to relate directly to the proportion of scenarios that result in acceptable performance of a system or strategy.

Flexibility can be defined as the extent to which a system can alter in the face of new information. When applied to water resources, this could mean plans that can branch in the future, deferring of options or staged implementation of interventions with an option to abandon at a point in the future. Real Options Analysis, Dynamic Adaptive Policy Pathways and Adaptation Tipping Points are all examples of tools or techniques that have been created to value such flexibility and reward it within the planning process (Roach, 2016).

Risk-based planning works by calculating the frequency and severity of system failures under multiple scenarios of supply and demand to provide an indication of the risk of crossing particular thresholds (Roach, 2016). It can therefore be useful in understanding risk versus cost appetite in decision makers (Borgomeo et al., 2014). It is currently used under conditions of deep uncertainty, with proponents arguing that threshold exceedance probabilities are much less sensitive that climate change probabilities (Dessai and Hulme, 2004).

A large amount of work has been done using all of these metrics or methods of assessing system or strategy performance. This is discussed in more detail in the section below which summarises top-down and bottom-up approaches.

Emerging and novel indicators related to public internet search terms may offer metrics with which to determine the social impacts of drought and drought interventions (Wilby et al., 2023), however, the use of these indicators within water resource management is as yet untested.

6.2. Top-down and bottom-up approaches

Decision making frameworks have tended to be divided up into top-down and bottom-up type approaches. Within large-scale water resource management planning, these terms have come to reference the treatment of uncertainty within the assessment. Top-down approaches tend to take downscaled climate change information, generate supply and demand projections and then test the vulnerability of the system at the bottom of the chain. Top-down approaches were originally designed to incorporate climate change projections into mitigation policy (Manous and Stakhiv, 2021). An example of such an analysis is the Intergovernmental Panel on Climate Change (IPCC) reports which take all GCMs and convert their outputs into impacts and messaging in an attempt to influence policy around mitigation and adaptation. This paradigm of using the GCM as a starting point in an assessment has permeated into water resource management planning, resulting in a number of tools and approaches that can be used within such a framework, but may be used to aid decision making. Such methods include Info-Gap decision theory, Robust Optimisation, Real Options Analysis.

Bottom-up approaches tend to focus initially on what factors the system or candidate strategy is most vulnerable to, which may or may not include climate information. Bottom-up planning approaches have been applied in a number of settings around the world, mainly in the United States through frameworks such as Robust Decision Making (Lempert et al., 2003), Decision Scaling (Brown et al., 2012), Climate-Risk Informed Decision Analysis (CRIDA) (Mendoza et al., 2018), and a number of tools exist to facilitate their implementation (Bennett et al., 2021). These decision making frameworks are yet to be applied in full in the UK, however such approaches have been applied to flood risk within the academic literature in the UK as a demonstrator of technical possibility (Prudhomme et al., 2010). The same approach has also been demonstrated within water resources through the Drought Vulnerability Framework (Environment Agency, 2015; UKWIR, 2017), however again, it is used more to identify vulnerability as a separate step within the management process, rather than being used as a central pillar of the decision making process.

These bottom-up approaches also tend to have a higher degree of stakeholder involvement and iteration. Among other differences, bottom-up approaches rely on testing a water resource system to a large range of uncertainty in order to elicit vulnerabilities. Such approaches are particularly attractive when the future is considered deeply uncertain, and an understanding of the system vulnerabilities is favoured over an understanding of the impact of scenarios on the system outputs (Lempert et al., 2003).

There are issues and benefits to using either approach that have been identified widely within academic literature (Chan et al., 2022). Top-down approaches tend to present a large range of uncertainty to decision makers, which critics argue may not be decision relevant and which contains inherent uncertainties that may not be obvious (Ludwig et al., 2014; Roach, 2016). Manous and Stakhiv (2021) argue that four important incompatibilities related to top-down approaches (the use of return periods and plausibility of climate scenarios, the linking of scenarios to non-stationary hydrology, the reliance on a probability distribution to calculate expected annual benefits and costs, and assumed discounting rates) are resolved through use of the CRIDA bottom-up approach. There is a weight of theoretical evidence that bottom-up approaches are substantially better suited to making system-level decisions about future planning (Lempert et al., 2003; Manous and Stakhiv, 2021) and that scenario-driven climate models provide insufficient information to be used at the scale required for water resource planning (Manous and Stakhiv, 2021). Where bottom-up approaches have been implemented, the limitations of the approach are not always clear to the reader, however there are known disadvantages related to the computation power required to generate and run scenarios (Groves and Lempert, 2007) and difficulties in analysing more than two dimensions on a response surface (Chan et al., 2022). The choice of which dimensions (climate variables) to include within an analysis may also have a significant effect on the outcome of the assessment (Culley et al., 2021). Whilst bottom-up studies have shown that the approach can be applied to complicated water resource supply systems (Turner et al., 2014), there appears to be a need to understand how such approaches might work within the UK decision context, as well as understanding the potential limitations and gaps in complementary tools. There are also potential issues (as with top-down planning) in reconciling different and interdependent spatial scales. There are some that argue that neither current bottom-up or top-down approaches fully appraise adaptation strategies, instead they define optimal water use and diagnose impacts (Ludwig et al., 2014).

Plausibility in relation to future scenarios is discussed in detail above. In relation to planning approaches, whilst plausibility is a key consideration for bottom-up approaches due to the importance of creating boundary conditions for the analysis, top-down approaches have historically not grappled with the issue of plausible future scenarios. Instead, top-down approaches tend to run a number of climate change model ensemble members through water resource models and generalise a result based on the percentile of ensemble members (such as the mean). Consequently, plausibility is often confounded within the assessment. Plausibility has historically been implemented within bottom-up assessments as a subjective statement, related to the level of theoretical understanding of the physical impacts of a variable, trends in driving data, climate models or paleo data and system sensitivity (Mendoza et al., 2018). Plausibility is also critical in robustness focussed approaches, and there are unanswered questions related to the plausibility of the scenarios, and whether they matter to the final decision. Zhang et al. (2021) demonstrated that removal of implausible models resulted in more robust interventions, however this is likely to be dependent on the nature of future changes at a given location. Lempert et al. (2003) argue that plausibility need only be assessed for those scenarios that a candidate strategy is not robust to. Further understanding is required to understand how differing approaches to plausibility could impact water resources assessments in the UK based on the nature of the projections available.

Combined, hybrid approaches have also been developed that integrate both bottom-up and top-down in some way (e.g. Girard et al. (2015)). UK water resource planning has also operated using a hybrid approach, starting from a point of top-down planning, incorporating aspects such as problem characterisation (Environment Agency, 2021a), or the most recent climate change guidance (Environment Agency, 2021c), which focusses effort within the modelling chain on aspects critical to the system under investigation. This modelling effort has focussed on quantifying baseline risk using historical or historically derived droughts (stochastic or synthetically produced using long term averages) and future risk from climate change, changes in demand or sustainability reductions. A number of scenarios are selected to explore the system response to changes (an increasing number over time with increased use of rapid and extendable modelling software packages such as GR6J and Pywr) before one scenario (or more if adaptive planning is being used) is selected against which to optimise future investments to. As of WRMP19, water companies have been able to follow an adaptive planning approach, whereby if their problem is sufficient that future scenarios cause different investment programmes to be proposed, decision points are identified where these plans diverge, allowing aspects of the future to reveal themselves before making a decision. Such a process contains elements of Robust Decision Making, predict-then-act, scenario-neutral (using the Drought Vulnerability Framework) and adaptive pathways, however due to the mixed approach, the principles of each are slightly altered to ensure they fit within a planning framework (Dessai and Darch, 2014).

There are issues in attempting to assign likelihood to future droughts (in addition to those discussed in relation to determining historical or stochastic drought likelihood). Whilst this has been undertaken for some datasets such as hydrological events within MaRIUS (Bell et al., 2018), there are issues related to non-stationarity, the influences of natural variability and climate signal, the length of records used within an analysis and the number of data points within available future climate sequences. Consequently, current planning approaches require identification of extreme droughts from within a baseline dataset, perturbation of these droughts using change factors, and then assessment of the impacts of these events as an extreme event (e.g. 1-in-500) plus climate change (Environment Agency, 2021b). This is fraught with difficulty, mainly due to the impacts of climate change making some events more or less severe depending on the water supply system, and a lack of understanding about whether these events are still representative of an extreme event (WRSE, 2021). Such issues are well known within the decision making literature, identified by (Manous and Stakhiv, 2021).

6.3. Choosing an approach

These approaches often attempt to force rational decision making onto the water resources problem in an objective way, however there is an argument that such an approach is time and resource intensive, and may be inappropriate for such problems (Siders and Pierce, 2021). Instead, the most suitable decision making approach may be found by understanding the decision making environment and tailoring the approach to suit (Siders and Pierce, 2021). Current water resource management planning guidelines (Environment Agency, 2021a) include freedom for water companies to emphasise or discount particular parts of the decision making framework, depending on their particular vulnerabilities (such an approach acts as a bridge between bottom-up and top-down methodologies), however broader understanding of the decision making environment is outside the scope of water company consideration.

If an objective decision is made on what a future framework should look like, there should be a focus on what the decision making environment is for water companies in England. Not only that, but also, on aspirations for components of the planning framework (e.g. the level of stakeholder engagement), the characteristics of what society values (e.g. watering gardens versus water for biodiversity in river environments) and how that value is calculated.

6.4. Future approaches

Due to the nature of the “problem” facing water resources, it is highly likely that approaches will emerge in the future that solve some part of this problem in a more satisfactory way than those implemented historically. Whilst it is difficult to speculate on the specifics of future decision making approaches, it is perhaps easier to comment on where technology may alter the characteristics of existing decision making methods. High resolution, large ensemble, transient future climate projections (Slingo *et al.*, 2022) may provide a large number of scenarios with which to test water supply systems in a transient way, which may permit an improved understanding of the timings, severity and frequency of future climate stressors. This type of approach (as implemented within Roach (2016)) may remove the need for the linear scaling of impacts back through time, which presents a significant uncertainty in the timing of when investments will be required (Environment Agency, 2021c). Temperature scaling has been implemented within HR Wallingford (2020) as a means of identifying when certain ensemble members reach particular levels of warming, which highlight the uncertainty in the timing of investments related to global warming levels. Such approaches were also explored through the enhanced Future Flows and Groundwater levels demonstrator projects (HR

Wallingford, in prep.), however a number of issues need to be overcome before such a framework is implemented. These include the use of standardised baseline datasets, improved methods to understand drought plausibility, enhanced climate model outputs and the incorporation of other sources of uncertainty as well as climate change. There is also a need for tools and approaches that compare different climate change projections datasets, examine the impacts of bias correction, drought metrics that are suitable for application to water supply system stress and screening tools for new datasets, as identified through the eFLaG user needs specification (HR Wallingford, 2021). The use of smart data and big data mining may also provide significant improvements in insights into demand patterns and provide feedback in real time.

The issue of the timing of climate change impacts, raised above, is one that also highlights issues between operational aspects and long term planning. There is little understanding of the implications of using linear scaling and the near-term risk associated with timing investments to cope with climate change in the next few years. It is also clear from issues such as 2018 demand levels and Covid-19 lockdown related demands, that despite current attempts, vulnerabilities still exist in the present day.

The likely increasing availability of computing power will allow more detailed systems or an larger number of scenarios to be run, or revolutionised by the development of quantum computing. Another critical aspect of the role of technology is the extent to which it is involved within the decision making process. Such a question is philosophical and has been discussed more widely as part of the machine ethics debate (e.g. Moor (2020)). Lempert *et al.* (2003) argue that the computer is a tool that should be used to synthesise quantities of information that the human mind cannot, provide an iterative process that allows for greater system testing, and present this in a way that enhances decision making. The incorporation of techniques such as Multi-Criteria Decision Analysis within decision making techniques has allowed for the scoring and weighting of different stakes within the decision, and the increasing potential of Artificial Intelligence may take this further by effectively allowing the computer to make (or at least suggest) a decision for us without full human reasoning. The question of how much agency a computer or human should have is a philosophical one that defines the decision making environment, reframing the involvement of stakeholders and the use of tools.

What we know

- **Multiple metrics exist for evaluating supply system performance. There is no agreement on which metric is preferred and the choice should depend on the decision context.**
- **Top-down approaches have historically dominated impact assessments.**
- **Bottom-up approaches combine an understanding of system vulnerability, plausibility and stakeholder engagement.**

- **Bottom-up approaches have been implemented outside the UK, as well as within the UK in water resource planning through the Drought Vulnerability Framework however never as a central pillar of the decision making framework.**
- **There are benefits and limitations to both approaches. Hybrid approaches exist that attempt to bridge the gap between the two. Current English water resource planning does this to some extent.**
- **Planning using current approaches has still resulted in vulnerabilities in the present day.**
- **A large number of barriers need to be addressed before moving towards a future, transient planning approach, including improvements in climate modelling and standardisation of baseline datasets.**
- There are planning issues related to the current approach of attempting to plan to a 1-in-500 year return period event. Current approaches also appear to be not fully exploring system vulnerabilities, particularly to multiple concurrent and / or interacting threats (e.g. 2018 drought demands and Covid-19 lockdown related demands)..

What we don't know

- **There are a lot of things we don't know in relation to water resources planning, hence the assumed condition of deep uncertainty.**
- **There is an absence of actionable information on the English planning decision context to enable a decision on the preferrable planning approach.**
- **The practical implications of moving to a different planning approach are not fully understood.**
- The plausibility of climate change scenarios are not fully visible to decision makers and planners.

What we should know

- **There should be an understanding of the decision context before choosing a planning approach.**
- **There should be an improved understanding of the practical implications of moving from one planning approach to another, and for incorporating this into guidance and planning.**
- **There should be an agreement on the philosophical principles of future planning approaches, particularly with regards to the involvement of stakeholders and the role of computers (including Artificial Intelligence and data mining) within decision making.**
- The plausibility of climate change scenarios should be visible to decision makers and planners to ensure the limitations of climate-related conclusions are known.

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References

Addor, N., Rohrer, M., Furrer, R. and Seibert, J. (2016) 'Propagation of biases in climate models from the synoptic to the regional scale: Implications for bias adjustment', Journal of Geophysical Research: Atmospheres, 121(5), pp. 2075–2089. Available at: https://doi.org/10.1002/2015JD024040.

Affinity Water (2022) Draft Water Resources Management Plan 2024.

Afzal, M. and Ragab, R. (2019) 'Drought risk under climate and land use changes: Implication to water resource availability at catchment scale', Water (Switzerland), 11(9). Available at: https://doi.org/10.3390/w11091790.

Afzal, M. and Ragab, R. (2020) 'How do climate and land use changes affect the water cycle? Modelling study including future drought events prediction using reliable drought indices', Irrigation and Drainage, 69(4), pp. 806–825. Available at: https://doi.org/10.1002/ird.2467.

Afzal, M., Vavlas, N. and Ragab, R. (2021) 'Modelling study to quantify the impact of future climate and land use changes on water resources availability at catchment scale', Journal of Water and Climate Change, 12(2), pp. 339–361. Available at: https://doi.org/10.2166/wcc.2020.117.

Alexander, L.V. and Jones, P.D. (2001) 'Updated Precipitation Series for the U.K. and Discussion of Recent Extremes', Atmospheric Science Letters, 1(2), pp. 142–150. Available at: https://doi.org/10.1006/asle.2001.0025.

Anderson, B., Manouseli, D. and Nagarajan, M. (2018) 'Estimating scenarios for domestic water demand under drought conditions in England and Wales', Water Science and Technology: Water Supply, 18(6), pp. 2100–2107. Available at: https://doi.org/10.2166/ws.2018.035.

Archer, D. (2003) 'Kielder Water: white elephant or white knight?', Tyne and tide: a celebration of the River Tyne, (January 1993), pp. 138–156.

Armstrong, M.S., Kiem, A.S. and Vance, T.R. (2020) 'Comparing instrumental, palaeoclimate, and projected rainfall data: Implications for water resources management and hydrological modelling', Journal of Hydrology: Regional Studies, 31(August), p. 100728. Available at: https://doi.org/10.1016/j.ejrh.2020.100728.

Ashton, C.H. and Hope, V.S. (2001) 'Environmental valuation and the economic level of leakage', Urban Water, 3(4), pp. 261–270. Available at: https://doi.org/10.1016/S1462-0758(01)00046-2.

Bakker, K.J. (2017) 'Privatizing water, producing scarcity: The Yorkshire drought of 1995', Environment: Critical Essays in Human Geography, 76(1), pp. 429–452. Available at: https://doi.org/10.4324/9781315256351-19.

Barnes, C. and Brierley, C. (2022) 'Indices to be used in comparison of UKCP18 and EuroCORDEX ensembles', (July). Available at: https://www.ucl.ac.uk/statistics/sites/statistics/files/ukcordex-indices_v08.pdf.

Bauer, P., Stevens, B. and Hazeleger, W. (2021) 'A digital twin of Earth for the green transition', Nature Climate Change, 11(2), pp. 80–83. Available at: https://doi.org/10.1038/s41558-021-00986-y.

Bell, V.A., Kay, A.L., Rudd, A.C. and Davies, H.N. (2018) 'The MaRIUS-G2G datasets: Grid-to-Grid model estimates of flow and soil moisture for Great Britain using observed and climate model driving data', Geoscience Data Journal, 5(2), pp. 63–72. Available at: https://doi.org/10.1002/gdj3.55.

Ben-Haim, Y. (2019) Info-Gap Decision Theory (IG), Decision Making under Deep Uncertainty. Available at: https:/doi.org/10.1007/978-3-030-05252-2_5.

Bennett, B., Devanand, A., Culley, S., Westra, S., Guo, D. and Maier, H.R. (2021) 'A modelling framework and R-package for evaluating system performance under hydroclimate variability and change', Environmental Modelling and Software, 139(February), p. 104999. Available at: https://doi.org/10.1016/j.envsoft.2021.104999.

Bennett, B.M. and Barton, G.A. (2018) 'The enduring link between forest cover and rainfall: a historical perspective on science and policy discussions', Forest Ecosystems, 5(1). Available at: https://doi.org/10.1186/s40663-017-0124-9.

Beswick, R.R., Oliveira, A.M. and Yan, Y. (2021) 'Does the Green Hydrogen Economy Have a Water Problem?', ACS Energy Letters, 6(9), pp. 3167–3169. Available at: https://doi.org/10.1021/acsenergylett.1c01375.

Birkinshaw, S.J., Bathurst, J.C. and Robinson, M. (2014) '45 years of non-stationary hydrology over a forest plantation growth cycle, Coalburn catchment, Northern England', Journal of Hydrology, 519(PA), pp. 559–573. Available at: https://doi.org/10.1016/j.jhydrol.2014.07.050.

Birks, C., Owen, M. and Arkell, B. (2023) 'London' s water resources : threat or opportunity', 33(1), pp. 95–97.

Bishop, C.H. and Abramowitz, G. (2013) 'Climate model dependence and the replicate Earth paradigm', Climate Dynamics, 41(3–4), pp. 885–900. Available at: https://doi.org/10.1007/s00382-012-1610-y.

Borgomeo, E., Hall, J.W., Fung, F., Watts, G., Colquhoun, K. and Lambert, C. (2014) 'Risk-based water resources planning: Incorporating probabilistic nonstationary climate uncertainties', Water Resources Research, 50(8), pp. 6850–6873. Available at: https://doi.org/10.1002/2014WR015558.

Bristol Water (2022) Bristol Water Draft Water Resources Management Plan 2024. Available at: https://7850638.fs1.hubspotusercontent-na1.net/hubfs/7850638/dWRMP2024 Technical Report FINALv2 REDACTED.pdf.

Brown, C., Ghile, Y., Laverty, M. and Li, K. (2012) ‘Decision scaling: Linking bottom-up vulnerability analysis with climate projections in the water sector’, Water Resources Research, 48(9), p. 2011WR011212. Available at: https://doi.org/10.1029/2011WR011212.

Brunner, M.I., Slater, L., Tallaksen, L.M. and Clark, M. (2021) ‘Challenges in modeling and predicting floods and droughts: A review’, Wiley Interdisciplinary Reviews: Water, 8(3), pp. 1–32. Available at: https://doi.org/10.1002/wat2.1520.

Chan, W., Shepherd, T., Facer-Childs, K., Darch, G., Arnell, N. and Van, K. (2022a) ‘Current and future risks of unprecedented UK droughts’, in EGU General Assembly. Vienna, Austria. Available at: https://doi.org/10.5194/egusphere-egu22-704.

Chan, W.C.H., Shepherd, T.G. and Arnell, N.W. (2022b) ‘Tracking the methodological evolution of climate change projections for UK river flows’, 0(0), pp. 1–24. Available at: https://doi.org/10.1177/03091333221079201.

Chun, K.P., Wheater, H.S. and Onof, C. (2013) ‘Comparaison des projections de sécheresse utilisant deux générateurs de données météorologiques du Royaume-Uni’, Hydrological Sciences Journal, 58(2), pp. 295–309. Available at: https://doi.org/10.1080/02626667.2012.754544.

Cooper, R.J., Melvin, T.M., Tyers, I., Wilson, R.J.S. and Briffa, K.R. (2013) ‘A tree-ring reconstruction of East Anglian (UK) hydroclimate variability over the last millennium’, Climate Dynamics, 40(3–4), pp. 1019–1039. Available at: https://doi.org/10.1007/s00382-012-1328-x.

Cotterill, D.F., Pope, J.O. and Stott, P.A. (2022) ‘Future extension of the UK summer and its impact on autumn precipitation’, Climate Dynamics [Preprint], (2021). Available at: https://doi.org/10.1007/s00382-022-06403-0.

Counsell, C.J.A. (2018) ‘End-to-end ensemble modelling for water resources planning under uncertainty’. Available at: https://doi.org/10.21954/ou.ro.0000dcf7.

CSIRO (2015) Climate Change in Australia Projections for Australia’s Natural Resource Management Regions: Technical Report.

Culley, S., Maier, H.R., Westra, S. and Bennett, B. (2021) ‘Identifying critical climate conditions for use in scenario-neutral climate impact assessments’, Environmental Modelling and Software, 136(December 2020), p. 104948. Available at: https://doi.org/10.1016/j.envsoft.2020.104948.

Darbandsari, P., Kerachian, R., Malakpour-Estalaki, S. and Khorasani, H. (2020) ‘An agent-based conflict resolution model for urban water resources management’, Sustainable Cities and Society, 57(January), p. 102112. Available at: https://doi.org/10.1016/j.scs.2020.102112.

Dawkins, L.C., Osborne, J.M., Economou, T., Darch, G.J.C. and Stoner, O.R. (2022) ‘The Advanced Meteorology Explorer: a novel stochastic, gridded daily rainfall generator', Journal of Hydrology, 607(February), p. 127478. Available at: https://doi.org/10.1016/j.jhydrol.2022.127478.

DEFRA (2019) 'Abstraction reform report', (May 2019), pp. 1–48. Available at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/801495/abstraction-reform-report.pdf.

Dessai, S. and Darch, G. (2014) 'Robust decision-making? Managing uncertainties in adapting water resource systems to a changing climate in England and Wales', in Climate Change Adaptation Manual: Lessons Learned from European and Other Industrialised Countries, pp. 208–212. Available at: https://www.scopus.com/inward/record.uri?eid=2-s2.0-85103711375&partnerID=40&md5=b5a616fa1ab7bd0488349ce461f5556a.

Dessai, S. and Hulme, M. (2004) 'Does climate adaptation policy need probabilities?', Climate Policy, 4(2), pp. 107–128. Available at: https://doi.org/10.1080/14693062.2004.9685515.

Diaz-Nieto, J. and Wilby, R.L. (2005) 'A comparison of statistical downscaling and climate change factor methods: Impacts on low flows in the River Thames, United Kingdom', Climatic Change, 69(2–3), pp. 245–268. Available at: https://doi.org/10.1007/s10584-005-1157-6.

Dobson, B., Coxon, G., Freer, J., Gavin, H., Mortazavi-Naeini, M. and Hall, J.W. (2020) 'The Spatial Dynamics of Droughts and Water Scarcity in England and Wales', Water Resources Research, 56(9). Available at: https://doi.org/10.1029/2020WR027187.

Doornkamp, J.C., Gregory, K.J. and Burn, A.S. (1980) Atlas of Drought in Britain 1975-76. Institute of British Geographers.

Douglas, J.R. (1988) 'Regulating the River Severn', Regulated Rivers: Research & Management, 2(3), pp. 309–321. Available at: https://doi.org/10.1002/rrr.3450020309.

Durant, M. (2015) Description of groundwater droughts in the UK: 1890 to 2015. Available at: http://nora.nerc.ac.uk/id/eprint/511707/1/OR15007.pdf.

Durant, M. and Counsell, C. (2020) 'Historic droughts inventory of drought references to water resources 1890-2015', [Data Collection]. Colchester, Essex: UK Data Service [Preprint]. Available at: https://doi.org/https://dx.doi.org/10.5255/UKDA-SN-853673.

Environment Agency (1995) 1995 Drought Report: A report on the severity of the drought of 1995 and its effect in the Severn and Trent catchments.

Environment Agency (2011) Water Demand for Energy Generation.

Environment Agency (2013) Quantifying the Impact of Water Company Drought Measures on Water Demand.

Environment Agency (2015) Understanding the performance of water supply systems during mild to extreme droughts.

Environment Agency (2020) 'Meeting our future water needs: a national framework for water resources', (March). Available at:

https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/872759/National_Framework_for_water_resources_main_report.pdf.

Environment Agency (2021a) Water Resources Planning Guideline. Available at: https://www.gov.uk/government/publications/water-resources-planning-guideline.

Environment Agency (2021b) Water Resources Planning Guideline Supplementary Guidance: 1 in 500. March 2021.

Environment Agency (2021c) Water Resources Planning Guideline Supplementary Guidance: Climate change. March 2021.

Environment Agency (2022) Review of England's emerging regional water resources plans. Available at: https://www.gov.uk/government/publications/review-of-englands-emerging-regional-water-resources-plans/review-of-englands-emerging-regional-water-resources-plans (Accessed: 23 January 2023).

Folland, C.K., Hannaford, J., Bloomfield, J.P., Kendon, M., Svensson, C., Marchant, B.P., Prior, J. and Wallace, E. (2015) 'Multi-annual droughts in the English Lowlands: A review of their characteristics and climate drivers in the winter half-year', Hydrology and Earth System Sciences, 19(5), pp. 2353–2375. Available at: https://doi.org/10.5194/hess-19-2353-2015.

Folland, C.K., Knight, J., Linderholm, H.W., Fereday, D., Ineson, S. and Hurrel, J.W. (2009) 'The summer North Atlantic oscillation: Past, present, and future', Journal of Climate, 22(5), pp. 1082–1103. Available at: https://doi.org/10.1175/2008JCLI2459.1.

Fowler, H.J., Blenkinsop, S. and Tebaldi, C. (2007) 'Linking climate change modelling to impacts studies: recent advances in downscaling techniques for hydrological modelling', International Journal of Climatology, 27(12), pp. 1547–1578. Available at: https://doi.org/10.1002/joc.1556.

Girard, C., Pulido-Velazquez, M., Rinaudo, J.D., Pagé, C. and Caballero, Y. (2015) 'Integrating top-down and bottom-up approaches to design global change adaptation at the river basin scale', Global Environmental Change, 34(September), pp. 132–146. Available at: https://doi.org/10.1016/j.gloenvcha.2015.07.002.

Glenis, V., Pinamonti, V., Hall, J.W. and Kilsby, C.G. (2015) 'A transient stochastic weather generator incorporating climate model uncertainty', Advances in Water Resources, 85, pp. 14–26. Available at: https://doi.org/10.1016/j.advwatres.2015.08.002.

Gohar, L.K., Lowe, J.A. and Bernie, D. (2017) 'The Impact of Bias Correction and Model Selection on Passing Temperature Thresholds', Journal of Geophysical Research: Atmospheres, 122(22), pp. 12,045-12,061. Available at: https://doi.org/10.1002/2017JD026797.

Goodchild, C.W. (2003) 'Modelling the impact of climate change on domestic water demand', Water and Environment Journal, 17(1), pp. 8–12. Available at: https://doi.org/10.1111/j.1747-6593.2003.tb00423.x.

Grecksch, K. and Stefán, Z. (2018) 'Drought, Water Scarcity and UK Businesses and Industries. An Exploratory Study into Challenges and Opportunities', SSRN Electronic Journal [Preprint]. Available at: https://doi.org/10.2139/ssrn.3256736.

Groves, D.G. and Lempert, R.J. (2007) 'A new analytic method for finding policy-relevant scenarios', Global Environmental Change, 17(1), pp. 73–85. Available at: https://doi.org/10.1016/j.gloenvcha.2006.11.006.

Guerreiro, S.B., Kilsby, C. and Fowler, H.J. (2017) 'Assessing the threat of future megadrought in Iberia', International Journal of Climatology, 37(15), pp. 5024–5034. Available at: https://doi.org/10.1002/joc.5140.

Hallegatte, S., Hourcade, J.C. and Ambrosi, P. (2007) 'Using climate analogues for assessing climate change economic impacts in urban areas', Climatic Change, 82(1–2), pp. 47–60. Available at: https://doi.org/10.1007/s10584-006-9161-z.

Hanlon, H.M., Bernie, D., Carigi, G. and Lowe, J.A. (2021) 'Future changes to high impact weather in the UK', Climatic Change, 166(3–4), pp. 1–23. Available at: https://doi.org/10.1007/s10584-021-03100-5.

Hannaford, J., Mackay, J., Ascott, M., Chitson, T., Cole, S., Counsell, C., Durant, M., Jackson, C.R., Kay, A.L., Lane, R.A., Moore, R., Parry, S., Rudd, A.C., Facer-Childs, K., Turner, S., Wallbank, J.R., Wells, S. and Wilcox, A. (2022) 'eFLaG: enhanced future FLows and Groundwater. A national dataset of hydrological projections based on UKCP18', Earth System Science Data Discussions, (February), pp. 1–40. Available at: https://doi.org/10.5194/essd-2022-40.

HR Wallingford (2020) 'Updated projections of future water availability for the third UK Climate Change Risk Assessment', (July), p. 147. Available at: https://www.ukclimaterisk.org/wp-content/uploads/2020/07/Updated-projections-of-future-water-availability_HRW.pdf.

HR Wallingford (2021) eFLaG : Enhanced Future Flows and Groundwater Levels: User Needs Specification and Research Requirements. Report Ref. FWR6277-RT001-R02-00. Available at: https://www.ceh.ac.uk/sites/default/files/2023-03/FWR6277-eFLaG_user_needs_spec-RT001-R02-00-final.pdf.

HR Wallingford (no date) eFLaG Demonstrator report. in prep.

Hutchins, M.G., Abesser, C., Prudhomme, C., Elliott, J.A., Bloomfield, J.P., Mansour, M.M. and Hitt, O.E. (2018) 'Combined impacts of future land-use and climate stressors on water resources and quality in groundwater and surface waterbodies of the upper Thames river basin, UK', Science of the Total Environment, 631–632, pp. 962–986. Available at: https://doi.org/10.1016/j.scitotenv.2018.03.052.

Kay, A.L., Watts, G., Wells, S.C. and Allen, S. (2020) 'The impact of climate change on U. K. river flows: A preliminary comparison of two generations of probabilistic climate projections', Hydrological Processes, 34(4), pp. 1081–1088. Available at: https://doi.org/10.1002/hyp.13644.

Kelder, T., Müller, M., Slater, L.J., Marjoribanks, T.I., Wilby, R.L., Prudhomme, C., Bohlinger, P., Ferranti, L. and Nipen, T. (2020) 'Using UNSEEN trends to detect decadal changes in 100-year precipitation extremes', npj Climate and Atmospheric Science 2020 3:1, 3(1), pp. 1–13. Available at: https://doi.org/10.1038/s41612-020-00149-4.

Keller, V.D.J., Tanguy, M., Prosdocimi, I., Terry, J.A., Hitt, O., Cole, S.J., Fry, M., Morris, D.G. and Dixon, H. (2015) ‘CEH-GEAR: 1 Km resolution daily and monthly areal rainfall estimates for the UK for hydrological and other applications’, Earth System Science Data, 7(1), pp. 143–155. Available at: https://doi.org/10.5194/essd-7-143-2015.

Knox, J.W., Kay, M.G., Hess, T.M. and Holman, I.P. (2020) ‘The challenges of developing an irrigation strategy for UK agriculture and horticulture in 2020: Industry and research priorities’, CAB Reviews: Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources, 15(50). Available at: https://doi.org/10.1079/PAVSNNR202015050.

Knox, J.W., Morris, J., Weatherhead, E.K. and Turner, A.P. (2000) ‘Mapping the financial benefits of sprinkler irrigation and potential financial impact of restrictions on abstraction: A case-study in Anglian Region’, Journal of Environmental Management, 58(1), pp. 45–59. Available at: https://doi.org/10.1006/jema.1999.0307.

Knutti, R., Masson, D. and Gettelman, A. (2013) ‘Climate model genealogy: Generation CMIP5 and how we got there’, Geophysical Research Letters, 40(6), pp. 1194–1199. Available at: https://doi.org/10.1002/grl.50256.

Kowalski, M. and Marshallsay, D. (2005) ‘Using measured microcomponent data to model the impact of water conservation strategies on the diurnal consumption profile’, Water Supply, 5(3–4), pp. 145–150. Available at: https://doi.org/10.2166/ws.2005.0094.

Kundzewicz, Z.W. and Stakhiv, E.Z. (2010) ‘Are climate models “ready for prime time” in water resources management applications, or is more research needed?’, Hydrological Sciences Journal, pp. 1085–1089. Available at: https://doi.org/10.1080/02626667.2010.513211.

Lafon, T., Dadson, S., Buys, G. and Prudhomme, C. (2013) ‘Bias correction of daily precipitation simulated by a regional climate model: A comparison of methods’, International Journal of Climatology, 33(6), pp. 1367–1381. Available at: https://doi.org/10.1002/joc.3518.

Lange, B., Holman, I. and Bloomfield, J.P. (2017) ‘A framework for a joint hydro-meteorological-social analysis of drought’, Science of the Total Environment, 578, pp. 297–306. Available at: https://doi.org/10.1016/j.scitotenv.2016.10.145.

Laux, P., Rötter, R.P., Webber, H., Dieng, D., Rahimi, J., Wei, J., Faye, B., Srivastava, A.K., Bliefernicht, J., Adeyeri, O., Arnault, J. and Kunstmann, H. (2021) ‘To bias correct or not to bias correct? An agricultural impact modelers’ perspective on regional climate model data’, Agricultural and Forest Meteorology, 304–305(April). Available at: https://doi.org/10.1016/j.agrformet.2021.108406.

LaVanchy, G.T., Kerwin, M.W. and Adamson, J.K. (2019) ‘Beyond “Day Zero”: insights and lessons from Cape Town (South Africa)’, Hydrogeology Journal, 27(5), pp. 1537–1540. Available at: https://doi.org/10.1007/s10040-019-01979-0.

Lempert, R., Popper, S. and Bankes, S. (2003) Shaping the Next One Hundred Years: New Methods for Quantitative, Long-Term Policy Analysis, Shaping the Next One Hundred Years: New Methods for Quantitative, Long-Term Policy Analysis. Available at: https://doi.org/10.7249/mr1626.

Loader, N.J., Young, G.H.F., McCarroll, D., Davies, D., Miles, D. and Bronk Ramsey, C. (2020) 'Summer precipitation for the England and Wales region, 1201–2000 ce, from stable oxygen isotopes in oak tree rings', Journal of Quaternary Science, 35(6), pp. 731–736. Available at: https://doi.org/10.1002/jqs.3226.

Lowe, J.A., Bernie, D., Bett, P.E., Bricheno, L., Brown, S.J., Calvert, D., Clark, R.T., Eagle, K.E., Edwards, T. and Fosser, G. (2018) 'UKCP18 science overview report', Met Office Hadley Centre: Exeter, UK, 2018(November 2018), pp. 1–73.

De Luca, P., Harpham, C., Wilby, R.L., Hillier, J.K., Franzke, C.L.E. and Leckebusch, G.C. (2019) 'Past and projected weather pattern persistence with associated multi-hazards in the British Isles', Atmosphere, 10(10), pp. 1–19. Available at: https://doi.org/10.3390/atmos10100577.

Ludwig, F., van Slobbe, E. and Cofino, W. (2014) 'Climate change adaptation and Integrated Water Resource Management in the water sector', Journal of Hydrology, 518(PB), pp. 235–242. Available at: https://doi.org/10.1016/j.jhydrol.2013.08.010.

Manous, J. and Stakhiv, E.Z. (2021) 'Climate risk-informed decision analysis (CRIDA): "top-down" vs "bottom-up" decision making for planning water resources infrastructure', Water Policy, 23, pp. 54–76. Available at: https://doi.org/10.2166/wp.2021.243.

Manouseli, D., Anderson, B. and Nagarajan, M. (2018) 'Domestic Water Demand During Droughts in Temperate Climates: Synthesising Evidence for an Integrated Framework', Water Resources Management, 32(2), pp. 433–447. Available at: https://doi.org/10.1007/s11269-017-1818-z.

Manouseli, D., Kayaga, S.M. and Kalawsky, R. (2017) 'Evaluation of water efficiency programs in single-family households in the UK: a case study', Water Supply, 17(6), pp. 1785–1792. Available at: https://doi.org/10.2166/WS.2017.071.

Mansour, M.M. and Hughes, A.G. (2018) Summary of results for national scale recharge modelling under conditions of predicted climate change. Available at: https://nora.nerc.ac.uk/id/eprint/521605/1/OR17026.pdf.

Maraun, D. (2016) 'Bias Correcting Climate Change Simulations - a Critical Review', Current Climate Change Reports, 2(4), pp. 211–220. Available at: https://doi.org/10.1007/s40641-016-0050-x.

Maraun, D., Shepherd, T.G., Widmann, M., Zappa, G., Walton, D., Gutiérrez, J.M., Hagemann, S., Richter, I., Soares, P.M.M., Hall, A. and Mearns, L.O. (2017) 'Towards process-informed bias correction of climate change simulations', Nature Climate Change, 7(11), pp. 764–773. Available at: https://doi.org/10.1038/nclimate3418.

Maraun, D., Truhetz, H. and Schaffer, A. (2021) 'Regional Climate Model Biases, Their Dependence on Synoptic Circulation Biases and the Potential for Bias Adjustment: A Process-Oriented Evaluation of the Austrian Regional Climate Projections', Journal of Geophysical Research: Atmospheres, 126(6). Available at: https://doi.org/10.1029/2020JD032824.

Marsh, T., Cole, G. and Wilby, R. (2007) 'Major droughts in England', Weather, 62(4), pp. 87–93.

Mendoza, G., Jeuken, A.B.M., Matthews, J.H., Stakhiv, E., Kucharski, J. and Gilroy, K. (2018) Climate Risk Informed Decision Analysis (CRIDA): collaborative water resources planning for an uncertain future, United Nations Educational, Scientific and Cultural Organization (UNESCO). Available at: https://unesdoc.unesco.org/ark:/48223/pf0000265895.

Met Office, Hollis, D., McCarthy, M., Kendon, M., Legg, T. and Simpson, I. (2018) 'HadUK-Grid gridded and regional average climate observations for the UK. Centre for Environmental Data Analysis'. Available at: http://catalogue.ceda.ac.uk/uuid/4dc8450d889a491ebb20e724debe2dfb.

Miller, K. and Yates, D. (2006) Climate change and water resources: A Primer for Municipal Water Providers. Available at: https://doi.org/10.1007/978-94-017-1051-0_2.

Moise, A., Wilson, L., Grose, M., Whetton, P., Watterson, I., Bhend, J., Bathols, J., Hanson, L., Erwin, T., Bedin, T., Heady, C. and Rafter, T. (2015) 'Evaluation of CMIP3 and CMIP5 Models over the Australian Region to Inform Confidence in Projections', Australian Meteorological and Oceanographic Journal, 65(1), pp. 19–53. Available at: https://doi.org/10.22499/2.6501.004.

Moon, H., Gudmundsson, L. and Seneviratne, S.I. (2018) 'Drought Persistence Errors in Global Climate Models', Journal of Geophysical Research: Atmospheres, 123(7), pp. 3483–3496. Available at: https://doi.org/10.1002/2017JD027577.

Moor, J.H. (2020) 'The mature, importance, and difficulty of machine ethics', Machine Ethics and Robot Ethics, pp. 233–236. Available at: https://doi.org/10.4324/9781003074991-21.

Murgatroyd, A. and Hall, J.W. (2020) 'The Resilience of Inter-basin Transfers to Severe Droughts With Changing Spatial Characteristics', Frontiers in Environmental Science, 8(December), pp. 1–22. Available at: https://doi.org/10.3389/fenvs.2020.571647.

Murphy, C., Wilby, R.L., Matthews, T., Horvath, C., Crampsie, A., Ludlow, F., Noone, S., Brannigan, J., Hannaford, J., McLeman, R. and Jobbova, E. (2020a) 'The forgotten drought of 1765–1768: Reconstructing and re-evaluating historical droughts in the British and Irish Isles', International Journal of Climatology, 40(12), pp. 5329–5351. Available at: https://doi.org/10.1002/joc.6521.

Murphy, C., Wilby, R.L., Matthews, T.K.R., Thorne, P., Broderick, C., Fealy, R., Hall, J., Harrigan, S., Jones, P., McCarthy, G., MacDonald, N., Noone, S. and Ryan, C. (2020b) 'Multi-century trends to wetter winters and drier summers in the England and Wales precipitation series explained by observational and sampling bias in early records', International Journal of Climatology, 40(1), pp. 610–619. Available at: https://doi.org/10.1002/joc.6208.

National Infrastructure Commission (2018) Preparing for a drier future. Available at: https://nic.org.uk/app/uploads/NIC-Preparing-for-a-Drier-Future-26-April-2018.pdf.

Parker, J.M. and Wilby, R.L. (2013) 'Quantifying Household Water Demand: A Review of Theory and Practice in the UK', Water Resources Management, 27(4), pp. 981–1011. Available at: https://doi.org/10.1007/s11269-012-0190-2.

Pearce, F. (1982) Watershed. The Water Crisis in Britain. London: Junction Books.

Pope, J.O., Brown, K., Fung, F., Hanlon, H.M., Neal, R., Palin, E.J. and Reid, A. (2021) 'Investigation of future climate change over the British Isles using weather patterns', Climate Dynamics, 58(9–10), pp. 2405–2419. Available at: https://doi.org/10.1007/s00382-021-06031-0.

Pribyl, K., Cornes, R.C. and Pfister, C. (2012) 'Reconstructing medieval April-July mean temperatures in East Anglia, 1256-1431', Climatic Change, 113(2), pp. 393–412. Available at: https://doi.org/10.1007/s10584-011-0327-y.

Prudhomme, C., Wilby, R.L., Crooks, S., Kay, A.L. and Reynard, N.S. (2010) 'Scenario-neutral approach to climate change impact studies: Application to flood risk', Journal of Hydrology, 390(3–4), pp. 198–209. Available at: https://doi.org/10.1016/j.jhydrol.2010.06.043.

Pullinger, M., Browne, A., Anderson, B. and Medd, W. (2013) Patterns of water: The water related practices of households in southern England, and their influence on water consumption and demand management.

Rahiz, M. and New, M. (2012) 'Spatial coherence of meteorological droughts in the UK since 1914', Area, 44(4), pp. 400–410. Available at: https://doi.org/10.1111/j.1475-4762.2012.01131.x.

Rahiz, M. and New, M. (2013) '21st Century Drought Scenarios for the UK', Water Resources Management, 27(4), pp. 1039–1061. Available at: https://doi.org/10.1007/s11269-012-0183-1.

Rey, D., Holman, I.P. and Knox, J.W. (2017) 'Developing drought resilience in irrigated agriculture in the face of increasing water scarcity', Regional Environmental Change, 17(5), pp. 1527–1540. Available at: https://doi.org/10.1007/s10113-017-1116-6.

Reyniers, N., Osborn, T.J., Addor, N. and Darch, G. (2022) 'Projected changes in droughts and extreme droughts in Great Britain strongly influenced by the choice of drought index', Hydrology and Earth System Sciences [Preprint], (March).

Ritchie, P.D.L., Harper, A.B., Smith, G.S., Kahana, R., Kendon, E.J., Lewis, H., Fezzi, C., Halleck-Vega, S., Boulton, C.A., Bateman, I.J. and Lenton, T.M. (2019) 'Large changes in Great Britain's vegetation and agricultural land-use predicted under unmitigated climate change', Environmental Research Letters, 14(11). Available at: https://doi.org/10.1088/1748-9326/ab492b.

Rittel, H.W.J. and Webber, M.M. (1973) 'Dilemmas in a General Theory of Planning Published by : Springer', Policy Sciences, 4(2), pp. 155–169.

Roach, T.P. (2016) Decision Making Methods for Water Resources Management Under Deep Uncertainty. Available at: https://ore.exeter.ac.uk/repository/bitstream/handle/10871/25756/RoachT.pdf?sequence=1&isAllowed=y.

Rodda, J.C. and Marsh, T.J. (2011) The 1975-76 Drought-a contemporary and retrospective review, National Hydrological Monitoring Programme series. Available at: www.ceh.ac.uk.

Rodina, L. (2019) ‘Water resilience lessons from Cape Town’s water crisis’, Wiley Interdisciplinary Reviews: Water, 6(6), pp. 1–7. Available at: https://doi.org/10.1002/WAT2.1376.

Rudd, A.C., Kay, A.L. and Bell, V.A. (2019) ‘National-scale analysis of future river flow and soil moisture droughts: potential changes in drought characteristics’, Climatic Change, 156(3), pp. 323–340. Available at: https://doi.org/10.1007/s10584-019-02528-0.

Rust, W., Holman, I., Bloomfield, J., Cuthbert, M. and Corstanje, R. (2019) ‘Understanding the potential of climate teleconnections to project future groundwater drought’, Hydrology and Earth System Sciences, 23(8), pp. 3233–3245. Available at: https://doi.org/10.5194/hess-23-3233-2019.

Serinaldi, F. and Kilsby, C.G. (2012) ‘A modular class of multisite monthly rainfall generators for water resource management and impact studies’, Journal of Hydrology, 464–465, pp. 528–540. Available at: https://doi.org/10.1016/j.jhydrol.2012.07.043.

Shaffrey, L. (2023) ‘Atmospheric Circulation, Dry Spells, and Drought in the UK’, This issue [Preprint].

Shepherd, T.G., Boyd, E., Calel, R.A., Chapman, S.C., Dessai, S., Dima-West, I.M., Fowler, H.J., James, R., Maraun, D., Martius, O., Senior, C.A., Sobel, A.H., Stainforth, D.A., Tett, S.F.B., Trenberth, K.E., van den Hurk, B.J.J.M., Watkins, N.W., Wilby, R.L. and Zenghelis, D.A. (2018) ‘Storylines: an alternative approach to representing uncertainty in physical aspects of climate change’, Climatic Change, 151(3–4), pp. 555–571. Available at: https://doi.org/10.1007/S10584-018-2317-9/FIGURES/1.

Siders, A.R. and Pierce, A.L. (2021) ‘Deciding how to make climate change adaptation decisions’, Current Opinion in Environmental Sustainability, 52, pp. 1–8. Available at: https://doi.org/10.1016/j.cosust.2021.03.017.

Slater, L.J., Anderson, B., Buechel, M., Dadson, S., Han, S., Harrigan, S., Kelder, T., Kowal, K., Lees, T., Matthews, T., Murphy, C. and Wilby, R.L. (2021) ‘Nonstationary weather and water extremes: A review of methods for their detection, attribution, and management’, Hydrology and Earth System Sciences, 25(7), pp. 3897–3935. Available at: https://doi.org/10.5194/HESS-25-3897-2021.

Slingo, J., Bates, P., Bauer, P., Belcher, S., Palmer, T., Stephens, G., Stevens, B., Stocker, T. and Teutsch, G. (2022) ‘Ambitious partnership needed for reliable climate prediction’, Nature Climate Change, 12(6), pp. 499–503. Available at: https://doi.org/10.1038/s41558-022-01384-8.

South East Water (2019) Drought Plan 2018-2023.

South West and Bournemouth Water (2019) Final Water Resources Management Plan August 2019. Available at: https://www.southwestwater.co.uk/siteassets/document-repository/environment/sww-bw-wrmp19---finalplan_aug2019.pdf.

South West and Bournemouth Water (2022) Final Drought Plan September 2022.

Spraggs, G., Peaver, L., Jones, P. and Ede, P. (2015) ‘Re-construction of historic drought in the Anglian Region (UK) over the period 1798–2010 and the implications for water resources and drought management’, Journal of Hydrology, 526, pp. 231–252.

Available at: https://doi.org/10.1016/J.JHYDROL.2015.01.015.

Sutcliffe, C., Knox, J. and Hess, T. (2021) ‘Managing irrigation under pressure: How supply chain demands and environmental objectives drive imbalance in agricultural resilience to water shortages’, Agricultural Water Management, 243(March 2020), p. 106484. Available at: https://doi.org/10.1016/j.agwat.2020.106484.

Svensson, C. and Hannaford, J. (2019) ‘Oceanic conditions associated with euro-atlantic high pressure and uk drought’, Environmental Research Communications, 1(10). Available at: https://doi.org/10.1088/2515-7620/ab42f7.

Taylor, V., Chappells, H., Medd, W. and Trentmann, F. (2009) ‘Drought is normal: the socio-technical evolution of drought and water demand in England and Wales, 1893-2006’, Journal of Historical Geography, 35(3), pp. 568–591. Available at: https://doi.org/10.1016/j.jhg.2008.09.004.

Thames Water (2022a) ‘Draft Drought Plan 2022’, (128). Available at: https://www.anglianwater.co.uk/about-us/our-strategies-and-plans/drought-plan/.

Thames Water (2022b) Draft Water Resources Management Plan. Technical Appendix F - Household Demand Forecasting. Available at: https://thames-wrmp.co.uk/assets/images/documents/technical-appendices/F-Household-Water-Demand-Forecast.pdf.

Thompson, V., Dunstone, N.J., Scaife, A.A., Smith, D.M., Slingo, J.M., Brown, S. and Belcher, S.E. (2017) ‘High risk of unprecedented UK rainfall in the current climate’, Nature Communications, 8(1), pp. 1–6. Available at: https://doi.org/10.1038/s41467-017-00275-3.

Todd, B. (2014) ‘Reconstructing long-term records of UK drought and analysis of variability: 1697-2013’, (September).

Turner, S.W.D., Marlow, D., Ekström, M., Rhodes, B.G., Kularathna, U. and Jeffrey, P.J. (2014) ‘Linking climate projections to performance: A yield-based decision scaling assessment of a large urban water resources system’, Water Resources Research, 50(4), pp. 3553–3567. Available at: https://doi.org/10.1002/2013WR015156.

UK Government (2021) ‘The England Trees Action Plan 2021-2024’, (May), pp. 1–36.

UKWIR (1995) Outage Allowances for Water Resource Planning. Report Ref. WRP-0001/B.

UKWIR (1997) A Review of Leakage. Updating Managing Leakage Methodology and Theory. Report Ref. No. 97/WM/08/10.

UKWIR (2007) Modelling the impact of restrictions on demand during drought. Report Ref. No. 07/WR/02/3. London.

UKWIR (2013) Understanding the impacts of drought restrictions. Report Ref. No. 14/WR/01/13. London.

UKWIR (2017) Drought Vulnerability Framework. Report Ref. 17/WR/02/12.

UKWIR (2021) Integrating UKCP18 with UKWIR Tools and Guidance: Review of Existing

Methods. Report Ref. No. 21/CL/04/19.

Wade, S., Sanderson, M., Golding, N., Lowe, J., Betts, R., Reynard, N., Kay, A., Stewart, L., Prudhomme, C., Shaffrey, L. and others (2015) Developing H++ climate change scenarios for heat waves, droughts, floods, windstorms and cold snaps.

Water UK (2016) Water resources long planning framework (2015-2065) Technical Report. Available at: https://www.water.org.uk/wp-content/uploads/2018/11/WaterUK-WRLTPF_Final-Report_FINAL-PUBLISHED-min.pdf.

Water UK (2022) Net Zero Routemap 2030: Summary for policymakers.

Weatherhead, E.K. and Howden, N.J.K. (2009) 'The relationship between land use and surface water resources in the UK', Land Use Policy, 26(SUPPL. 1), pp. 243–250. Available at: https://doi.org/10.1016/j.landusepol.2009.08.007.

West, H., Quinn, N. and Horswell, M. (2022) 'The Influence of the North Atlantic Oscillation and East Atlantic Pattern on Drought in British Catchments', Frontiers in Environmental Science, 10(March), pp. 1–18. Available at: https://doi.org/10.3389/fenvs.2022.754597.

Wetter, O., Pfister, C., Werner, J.P., Zorita, E., Wagner, S., Seneviratne, S.I., Herget, J., Grünewald, U., Luterbacher, J., Alcoforado, M.J., Barriendos, M., Bieber, U., Brázdil, R., Burmeister, K.H., Camenisch, C., Contino, A., Dobrovolný, P., Glaser, R., Himmelsbach, I., et al. (2014) 'The year-long unprecedented European heat and drought of 1540 - a worst case', Climatic Change, 125(3–4), pp. 349–363. Available at: https://doi.org/10.1007/S10584-014-1184-2/FIGURES/3.

Wilby, R.L. (2010) 'Evaluating climate model outputs for hydrological applications', Hydrological Sciences Journal, 55(7), pp. 1090–1093. Available at: https://doi.org/10.1080/02626667.2010.513212.

Wilby, R.L., Clifford, N.J., De Luca, P., Harrigan, S., Hillier, J.K., Hodgkins, R., Johnson, M.F., Matthews, T.K.R., Murphy, C., Noone, S.J., Parry, S., Prudhomme, C., Rice, S.P., Slater, L.J., Smith, K.A. and Wood, P.J. (2017) 'The "dirty dozen" of freshwater science: detecting then reconciling hydrological data biases and errors', Wiley Interdisciplinary Reviews: Water, 4(3). Available at: https://doi.org/10.1002/WAT2.1209.

Wilby, R.L., Fenn, C.R., Wood, P.J., Timlett, R. and Lequesne, T. (2011) 'Smart licensing and environmental flows: Modeling framework and sensitivity testing', Water Resources Research, 47(12), pp. 1–15. Available at: https://doi.org/10.1029/2011WR011194.

Wilby, R.L., Murphy, C., O'Connor, P., Thompson, J.J. and Matthews, T. (2023) 'Novel indicators to inform drought management and water planning', This issue [Preprint].

Wilks, D.S. and Wilby, R.L. (1999) 'The weather generation game: A review of stochastic weather models', Progress in Physical Geography, 23(3), pp. 329–357. Available at: https://doi.org/10.1177/030913339902300302.

Wilson, R., Miles, D., Loader, N.J., Melvin, T., Cunningham, L., Cooper, R. and Briffa, K. (2013) 'A millennial long March-July precipitation reconstruction for southern-central England', Climate Dynamics, 40(3–4), pp. 997–1017. Available at:

https://doi.org/10.1007/s00382-012-1318-z.

Woodhouse, C.A., Lukas, J.J., Morino, K., Meko, D.M. and Hirschboeck, K.K. (2016) 'Using the Past to Plan for the Future - The Value of Paleoclimate Reconstructions for Water Resource Planning', in K.A. Miller, A.F. Hamlet, D.S. Kenney, and K.T. Redmond (eds) Water Policy and Planning in a Variable and Changing Climate. Taylor & Francis, pp. 161–182.

WRSE (2021) 'Method Statement : Climate Change – Supply Side Methods', (August).

Yang, Y., Lerner, D.N., Barrett, M.H. and Tellam, J.H. (1999) 'Quantification of groundwater recharge in the city of Nottingham, UK', Environmental Geology, 38(3), pp. 183–198. Available at: https://doi.org/10.1007/s002540050414.

Zhang, R., Cheng, L., Liu, P., Huang, K., Gong, Y., Qin, S. and Liu, D. (2021) 'Effect of GCM credibility on water resource system robustness under climate change based on decision scaling', Advances in Water Resources, 158(September), p. 104063. Available at: https://doi.org/10.1016/j.advwatres.2021.104063.

M: Models and data in use within the water sector and their relevance to drought management, planning and practice

Gemma Coxon[1], Francesca Pianosi[2] and Doris E. Wendt[2]

[1] School of Geographical Sciences, University of Bristol.

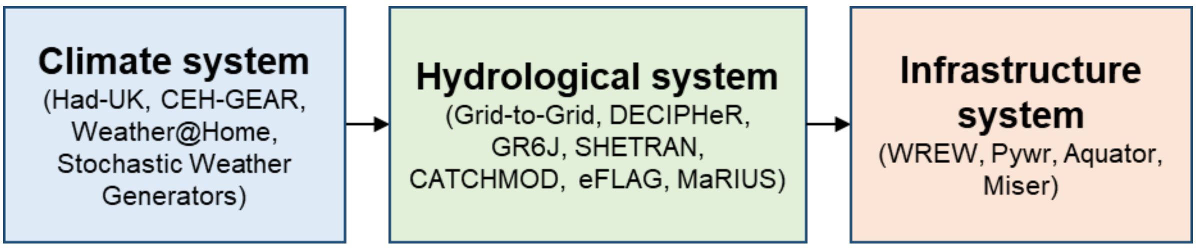
[2] School of Civil, Aerospace and Design Engineering, University of Bristol.

Overview

ABOUT THE REVIEW

Quantitative assessments of hydro-meteorological droughts provide key information to support drought management decisions and water resource planning. Over time, drought assessments have become increasingly complex with a significant increase in the models and data in use across the water sector.

What is covered: Our review assesses this breadth of climate, hydrology and water resource system datasets and models in use across the UK water sector. We evaluate these models and datasets against a set of four core requirements for drought management to assess their capabilities to address key knowledge gaps on current and future droughts. The four requirements include: the ability to take on a long-term view under change, the spatial coherence at national scale, the ability to consider a breadth of scenarios and the inclusion of high impact, low likelihood scenarios.
What is not covered: Our review does not assess groundwater models or datasets.



KEY FINDINGS

What can we do	What we cannot do
✓ We can produce long term, spatially coherent, national-scale climate, hydrological and water resource projections for drought management, planning and assessment.	× None of the datasets and models meet all the core requirements and all suffer from similar deficiencies. × We lack a wide range of climate and socio-economic scenarios for long-term drought projections, and we lack the ability to assess high impact, low likelihood extreme droughts.

RECOMMENDATIONS

Structured approaches are needed to **prioritise efforts for improvement of datasets and models** through a better understanding of key sensitivities of drought drivers and metrics.

Structured approaches are needed to **document, evaluate and compare datasets and models** through sharing models/data and better modelling protocols to ensure comparability.

A **greater breadth of scenarios** is needed alongside modelling frameworks to assess them through storyline approaches, greater sampling of uncertainties and consideration of both climate and socio-economic change in future drought assessments.

1. Abstract

Quantitative assessments of hydro-meteorological droughts provide key information to support drought management decisions and water resource planning. Over time, drought assessments have become increasingly complex with a significant increase in the models and data in use across the UK water sector. This review assesses the datasets and models most commonly used across the UK water sector, focusing in particular on climate datasets, hydrological and water resource system models, and hydrological projections. We evaluate these models and datasets against a set of four core requirements for drought management: the ability to take on a long-term view under change, the spatial coherence at national scale, the ability to consider a breadth of scenarios and the inclusion of high impact, low likelihood scenarios. We find that datasets and models all have varying strengths and weaknesses.

Crucially, none of the datasets/models meets all the core requirements. We conclude that more structured approaches are required to prioritise efforts for improvement of datasets and models, including a better understanding of key sensitivities of drought drivers and metrics. Furthermore, we need more structured approaches to document, evaluate and compare existing and future datasets and models. Finally, we identify and discuss key gaps in our ability to assess a breadth of scenarios and high impact, low likelihood scenarios, and stress the need for more strategic research in this area

2. Background and Rationale

A reliable water supply is often taken for granted in the UK. However, there is increasing pressure on water supplies. By 2050, the UK population is projected to rise by 10% from 67million to 74million, with demand expected to increase by up to 9% to supply the growing population with water for drinking, food supply and energy production (Environment Agency, 2020). In contrast, the headline findings from the new high resolution UK Climate Projections (UKCP18) forecast hotter and drier summers that will exacerbate drought risk (Murphy et al., 2018). These changes are projected to lead to frequent water shortages across the UK by 2050, with projections estimating more than 3 billion litres of additional water a day required to ensure supply (Environment Agency, 2020). Given the projected increasing pressure on water bodies, predicting future droughts and their impacts on water supply systems is vital to test their resilience and to support critical planning decisions for future water supply infrastructure (National Infrastructure Commission, 2018).

Droughts are a complex hazard that can occur in different parts of the hydrological system and have many definitions in the literature. In this review, we define drought events as a sustained period of below normal water availability (Tallaksen and Lanen, 2004).

Meteorological droughts refer to a precipitation deficiency, soil moisture droughts refer to a deficit of soil moisture and hydrological droughts are deficits in surface and/or subsurface water (Van Loon, 2015). These different types of droughts are heavily interlinked. Drought events often begin as meteorological droughts, which then lead to shortages in soil moisture, recharge, discharge, and groundwater storage that propagate through the hydrological cycle. Climate models aim to assess or forecast meteorological droughts at regional, national, or global scale while propagation of meteorological droughts through the hydrological cycle is often represented in hydrological models. One of the difficulties in modelling drought propagation is the delayed occurrence from a meteorological drought to a soil moisture drought followed by a hydrological drought. The severity of a meteorological drought event, catchment characteristics, local and regional water storage, subsurface storage and release and antecedent conditions are all important when it comes to the timing of developing soil moisture and hydrological droughts (Van Loon et al., 2016).

Over time, drought assessments have become increasingly sophisticated to capture these complex hazards and how they might be changing. They have progressed from scaling river flows using simple change factors based on a single Global Circulation Model (Arnell, 2011) to scaling historic rainfall timeseries using large ensembles of change factors encompassing different climate models and emission scenarios (e.g. Christierson et al., 2012; HR Wallingford, 2015) to running hundreds of years of climate model or stochastic weather generator timeseries through groundwater, hydrological and water resource system models (Dobson et al., 2020; Hannaford et al., 2022). As drought assessments have become increasingly sophisticated, there has been a significant increase in the models and data available covering a wide range of spatial/temporal scales, physical processes, availability (i.e. open-source or not), levels of uncertainty and purposes. Thus, each model and dataset has their own advantages and disadvantages.

While there is an extensive range of models and data in use within the water sector, there is a lack of guidance of which models and data to use for drought management and planning. Furthermore, there is little assessment of the capabilities of these models and data to address key knowledge gaps on current and future droughts. This review will address this challenge by reviewing models and data in use within the UK water industry and/or academic community and their relevance to drought management, planning and practice.

We will review the peer-reviewed academic literature and relevant ‘grey’ literature (e.g. water resource company plans, water resource management plan guidance produced by the EA, consultancy reports) to address the following questions:

1. Which data and models are currently available for use in water resources and drought planning? What are their strengths/weaknesses?
2. How uniform/fragmented are current approaches? Are there obvious gaps in terms of the processes they include, temporal/spatial resolution, availability?
3. What are current capabilities for addressing key knowledge gaps on current and future droughts? What new data/modelling capabilities are needed?

This review is structured into six sections beginning with an overview of the drought modelling chain and then leading into an evaluation of a number of climate projections, hydrological models/projections and water resource system models. Finally, we end the report with a series of recommendations that provide an outlook of how we can best use our current suite of data and models for drought management and what new data and modelling capabilities are needed to address the UK's future drought challenges.

3. Drought Modelling Chain

Drought management and planning requires a clear understanding of current drought variability and how the frequency, severity and duration of hydrological droughts might change in the future. To address this challenge, drought assessments typically comprise of three core elements: the climate, hydrological and infrastructure system. These core elements represent the complex interactions of water fluxes between the atmosphere, the land surface and sub-surface, and water infrastructures that occur during droughts. This is summarized in Figure 1 which demonstrates the typical flow of data between these systems, alongside the models and data analysed in this review.

The first component of the drought modelling chain is the **climate system**. Typically the most important climate variables for drought assessment are precipitation and potential-evapotranspiration (PET) as these are the key climatic drivers of hydrological droughts. Timeseries of precipitation and potential-evapotranspiration can be derived from meteorological observations, but this will provide a very limited set of meteorological drought events and does not account for how climate change may impact the duration, severity and frequency of future droughts in the hydrological system. Consequently, climate models and stochastic weather generators are used to provide larger ensembles of meteorological droughts. These datasets are evaluated in Section 3, which covers climate data for drought planning including rainfall and PET observational data products, UK Climate Projections 2018, Weather@Home climate projections and outputs from stochastic weather generators.

Outputs from the climate system (timeseries of precipitation and potential evapotranspiration) are then used as inputs for the second component of the drought modelling chain: the **hydrological system.** The hydrological system represents terrestrial water fluxes across the surface and sub-surface, which are codified into hydrological and groundwater models. Alongside inputs from the climate system, these models often require soils, geology and landcover data to define parameter values, and/or observed river flows and groundwater levels for calibration of parameters. Model outputs are river flows and groundwater levels. These models and the model output datasets they provide are evaluated in detail in Section 4, which covers six hydrological models (e.g. Grid-to-Grid, DECIPHeR, GR4J) and three sets of hydrological projections. Groundwater models are not evaluated in this review but are reviewed elsewhere as part of the EA drought reviews.

The third component of the drought modelling chain is the **infrastructure system**, which includes infrastructure for water supply and distribution, such as dams, diversions, canals etc. The movement of water through these components is simulated by water resource systems models. These models are typically forced by hydrological inputs such as river flows (either observed or simulated by hydrological models) and socio-economic-legal inputs such as water demands or minimum environmental flow requirements. They also need data about the system properties, for example the maximum capacity of reservoirs and canals, for their parameterization. They can be used in two different ways: (i) in simulation mode, to estimate the effects of given operation rules on the system performances, measured for example by supply reliability or energy cost; or (ii) in optimisation mode, to identify the best operations to maximise either of those performances. These models (Aquator, MISER, Pywr, WREW) are evaluated in Section 4. Note that this distinction between hydrological and water resource system models, while relatively clear-cut in current practice, may become less relevant in the future as new research developments will deliver hydrological models that explicitly represent water infrastructure and their operation.

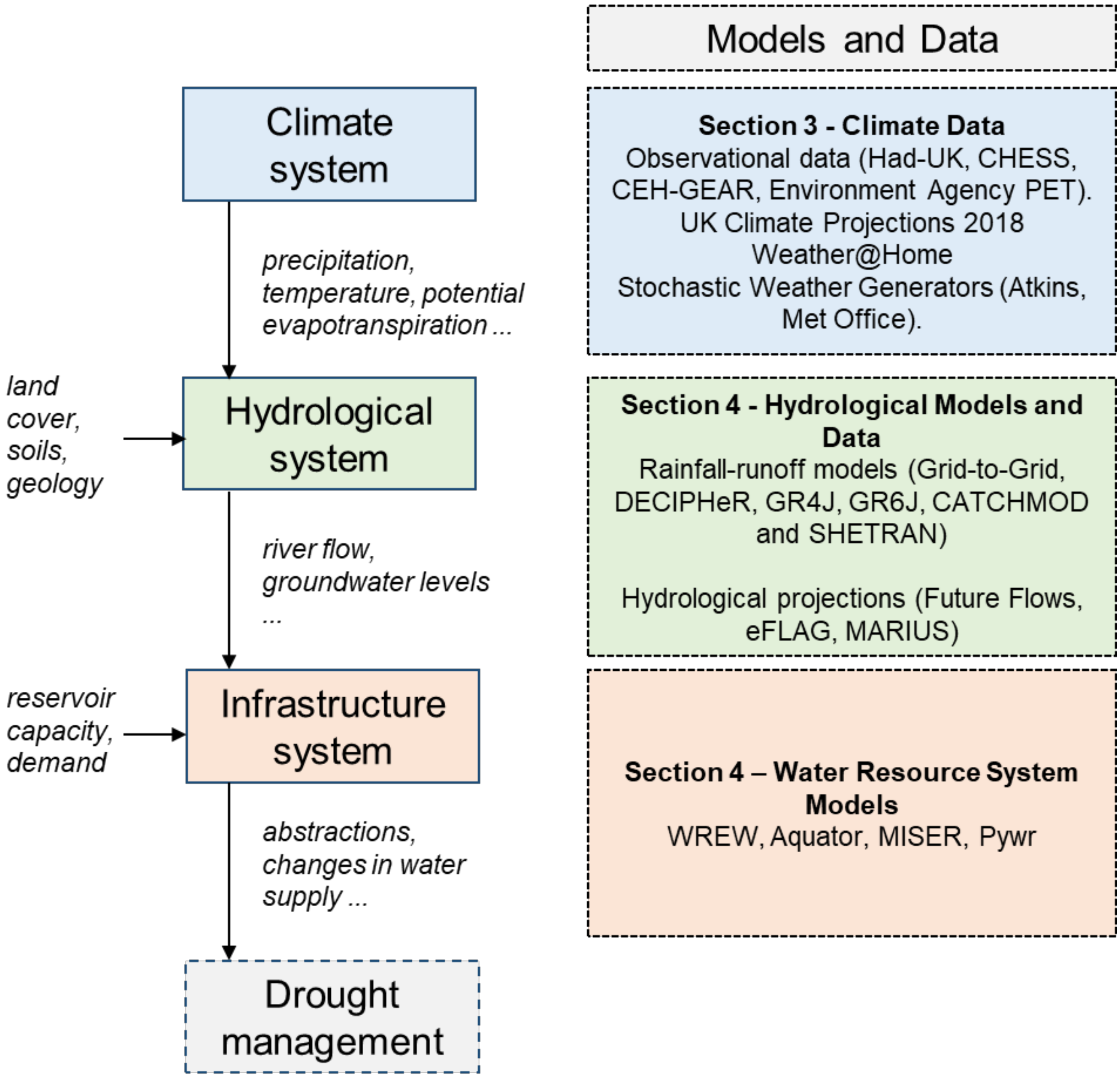


Figure 1. Flow diagram showing the drought modelling chain, alongside the models and data included in this review

There are several core principles central to drought management and planning in the UK that link into national planning guidelines (Atkins, 2020; Environment Agency, 2022). These are important for the selection of suitable climate, hydrological and water resource datasets and models. Firstly, given the significant long-term investments required for any new water infrastructure, drought assessments should take a **long-term view under change** and consider drought risks under the current climate as well as climate and socio-economic scenarios to the end of the century. Secondly, drought assessments should consider a **breadth of scenarios** (e.g. different climate and socio-economic scenarios) that encompass a wide range of plausible drought conditions. Thirdly, drought assessments should include an appraisal of **high impact, low likelihood extreme**

droughts to ensure the resilience of our water supply systems. Finally, given the significant spatial scales of droughts and consideration of regional water transfers, datasets that offer **spatial coherence at national- scale** are desirable.

These core principles mean that drought assessments need to consider a wide range of factors for each component of the modelling chain, including:

Long-term view under change

Temporal resolution and coverage i.e. timestep and the time period(s) covered.

Model calibration and evaluation; i.e. is the model an adequate representation of the real-world system and the task it is intended for?

Spatial coherence at national-scale

Spatial coverage and resolution; i.e. grid size and point-scale to continental-scale data

Spatial coherency; i.e. coherence between different geographical locations.

High impact, low likelihood extreme droughts

Extremes; i.e. how do we quantify extreme events and ensure our water systems are robust to them?

Model calibration and evaluation; i.e. can the models represent extremes adequately?

Breadth of scenarios

Uncertainty quantification; i.e. do the scenarios cover a wide range of plausible future drought events?

Uncertainty propagation; i.e. how much uncertainty is introduced at each stage of the modelling chain and how does it propagate into the next system?

4. Climate System

There are a wide range of climate data and models in use across the UK water sector for drought planning, management and assessment. In this review, we focus on rainfall and potential evapotranspiration timeseries as the key climate variables for drought assessment and the following observational and climate model data:

1. Observational rainfall and potential evapotranspiration data products
2. UK Climate Projections 2018 (UKCP18)

3. MaRIUS Weather@Home projections

4. Stochastic weather generator outputs

We start this section with a short description of the climate data listed above and then discuss their strengths and weaknesses. The key characteristics of the climate data products described in Section 4.1 are summarised in the appendix in tables A1 and A2.

4.1. Climate data for drought management and planning

Observational rainfall and PET data are key timeseries to improve our understanding of the meteorological drivers of past droughts, to calibrate and evaluate hydrological models, and for stress-testing water supply system models against historic droughts. There are a wide range of observational climate data available (see Table A1), included gridded observational climate data (e.g. CEH-GEAR, Had-UK, CHESS, Environment Agency PET) and catchment averaged observational climate data (e.g. CAMELS-GB). The gridded observational products are provided on a 1km grid, typically at a daily timescale (except for CEH- GEAR1hr). The gridded daily rainfall products are available from the 1890s to the present. Both of the daily gridded rainfall products are derived from the archive of UK weather observations held at the Met Office and then interpolated to a 1km grid either using nearest neighbour (CEH-GEAR) or inverse distance weighting (Had-UK). The hourly rainfall data (CEH-GEAR1hr) are derived by temporally disaggregating the CEH-GEAR daily rainfall dataset using a national database of hourly rain gauge observations. Most of the daily PET gridded products are available from the 1960s to 2020s and typically calculated using the Penman-Monteith equation for a well-watered grass surface (both with and without an interception correction). The one exception is the Historic Gridded PET dataset which is available from 1890-2015 and calculated using temperature data only. All the PET datasets use different underlying meteorological datasets at different resolutions. Observational data products for rainfall and PET cover either the UK, Great Britain or England and Wales, facilitating their use in national-scale analyses.

The UK Climate Projections 2018 (UKCP18) are the newest set of climate projections for the UK (Lowe et al., 2018). They are already widely used across the UK water sector including a number of recent publications (e.g. Hannaford et al., 2022; Lane et al., 2022; Kay et al., 2021; Chan et al., 2022) and water company water resource management plans (e.g. (Wessex Water, 2023; Thames Water, 2022). UKCP18 includes a suite of climate data products including probabilistic, global, regional, and local projections all derived from climate models. The probabilistic projections provide probabilistic changes of future climate based on uncertainty in emission scenarios (RCP2.6, RCP4.5, RCP6.0, RCP8.5 and SRESA1B) and key processes in climate models. The global (60km, daily), regional (12km, daily) and local (2.2km, hourly) projections provide gridded climate model outputs for 200 years (1900-2100), 100 years (1981-2080) and three time slices (1981-2000, 2021-2040 and 2061-2080) respectively. They are available for one single emissions scenario (RCP8.5).

While the regional and local projections include outputs from a single climate model with 12 ensemble members, the global projections include outputs from the Met Office Global Model and CMIP5 climate models (28 ensemble members in total).

The MaRIUS Weather@Home climate projections were developed as part of the UK Droughts and Water Scarcity Programme (Guillod et al., 2017). They have been widely used across the UK water sector, specifically assessing the impact of future climate change on the water supply system (Dobson et al., 2020; Murgatroyd et al., 2022), agriculture (Chengot et al., 2023) and energy sector (Byers et al., 2020). They are also used by the Environment Agency and Ofwat as part of the RAPID (Regulators' Alliance for Progressing Infrastructure Development) programme to assess the development of new water infrastructure (Ofwat, 2022). The MaRIUS Weather@Home climate projections provide gridded climate model output (25km) and include sets of 100 daily timeseries generated for three time slices (1900- 2006, 2020-2049 and 2070-2099). For each timeslice, uncertainty in transient climate response is considered by sampling a range of five sea surface temperature (SST) warming patterns derived from the CMIP5 (Coupled Model Intercomparison Project Phase 5) models. Only one emission scenario (RCP8.5) is included.

Stochastic weather generator outputs have been developed and produced by several groups, and are typically used to simulate very large ensembles of precipitation and PET to inform the calculation of extreme meteorological and/or hydrological drought events (e.g. 1 in 200 or 500 year drought event). The regional stochastics dataset provided by Atkins is currently the most commonly used stochastics dataset across the UK water sector, particularly in water company resource management plans (e.g. Thames Water, 2022; Wessex Water, 2023). It implements stochastic multisite weather generators (Serinaldi and Kilsby, 2012) driven by a range of climate drivers (e.g. North Atlantic Oscillation, sea surface temperatures, Atlantic Multidecadal Oscillation). The dataset consists of 400 stochastic replicates of precipitation and PET for 1950-1997. Spatially coherent precipitation timeseries for each region are generated for over 200 locations across England and Wales. Another stochastics weather generator in use in the UK water sector is the Advanced Meteorology Explorer (AME) developed by the UK Met Office in collaboration with Anglian Water (Dawkins et al., 2022; Anglian Water, 2022). The AME is based on hidden Markov models and copulas. It allows for the simulation of spatially and temporally coherent, synthetic daily rainfall data on a high-resolution grid. The AME framework has been used to generate 1000 alternative realisations of the time period 1914-2018 (105 years) on a 5 km grid over the Greater Anglian region using daily rainfall data from 39 sites.

4.2. Strengths and weaknesses of current climate data products

The climate datasets described above meet different needs for drought management, assessment, and planning, and thus have varying strengths and weaknesses (see Table 1).

All the climate model outputs (i.e. UKCP18 and MaRIUS) meet the need for ensuring a **long view under change** in drought assessments. While there are small differences in time periods available, both the suite of UKCP18 products and MaRIUS Weather@Home projections enable drought risks to be assessed under the current climate and future climate to the end of the century. Observational datasets clearly enable drought risks to be assessed under the current climate but not future climate scenarios. This is the same as the stochastics datasets as the outputs are based on a baseline climate *without climate change*. The observational data and stochastics datasets have been used for long term planning under climate change by applying change factors (i.e. percentage changes in precipitation and PET from a baseline climate to a future climate, typically derived from climate models) to observed and/or stochastic time series (see for example Anglian Water, 2022 and water resources planning guidelines from the Environment Agency). However, perturbing historic weather sequences using a change factor approach means that critical changes in drought frequency, duration and severity are not captured (Watts et al., 2015) and thus multi-year droughts that are projected to occur more frequently under climate change will not be represented. This approach also assumes that biases are constant over time and that current spatial patterns will remain the same in the future (Fowler et al., 2016), which may be incorrect and lead to plausible future drought conditions being overlooked. While change factor approaches are easy to apply, they should be used with caution and in acknowledgement of these assumptions.

Only some of the climate model outputs enable a **breadth of scenarios** to be considered. The gridded climate model outputs provide a set of future climate outputs for only RCP8.5. RCP8.5 represents the high-emissions scenario and is only one of the wide range of possible future emission scenarios. The UK Met Office Hadley Model is also hotter than other CMIP5 models. This has implications for water resources planning as these model outputs will show a greater impact on river flows due to the much hotter and drier climate present in these emission scenarios and climate model. The UKCP18 probabilistic projections include the full range of emission scenarios but have limitations as they provide percentage changes of future climate that can only scale historic weather sequences (see critique of change factors in previous paragraph). The global projections provide 13 outputs from CMIP5 models in addition to the UK Met Office Hadley Model and thus capture some of the uncertainty in choice of climate model. However, these are at a coarse spatial resolution (60km) for UK drought management and would need downscaling.

For the assessment of **high impact, low likelihood extreme droughts**, the stochastics datasets provide the largest climate dataset to generate estimates of extreme events. These datasets were specifically developed to meet the requirement of assessing more extreme events, such as 1 in 200 or 1 in 500 year drought events. However, stochastic weather generators are heavily reliant on their input data and the choice of time period that is modelled. As an example, by generating 400 stochastic replicates of the weather from 1950- 1997, the Atkins stochastics dataset may miss significant historic droughts for some regions of the UK that occurred in the earlier half of the 20th century (for example the 1933-35 drought) or emphasise particularly extreme droughts in the record (e.g. 1976). The Atkins stochastic timeseries also provide monthly outputs (i.e. total monthly rainfall) which need to be downscaled to daily for water resources planning. Furthermore it is worth noting that the stochastics outputs generate multiple timeseries of 48 or 105 years of data, but these timeseries are not independent and therefore extreme events should not be derived using all years of data. While there are limitations of the stochastics datasets, they do provide a much larger dataset to assess possible drought scenarios compared to the gridded climate outputs and are a useful tool for stress-testing water resource systems. Despite the available climate products and datasets, a comprehensive assessment of extreme meteorological drought events for the gridded climate outputs is lacking. Guillod et al., (2017) assess return time periods of 5-50 years for low rainfall accumulated over 1-4 hydrological years but there has been little further assessment of low likelihood extreme droughts in the MaRIUS Weather@Home dataset or in the UKCP18 dataset.

Finally, most of the climate products enable **spatially coherent national-scale** drought assessments except for the UKCP18 probabilistic projections and the stochastics datasets. The Atkins dataset is spatially coherent within regions but not across regions, and only covers England and Wales. The Met Office stochastics dataset currently only covers the Greater Anglian region of England. In terms of spatial resolution, the UKCP18 global projections at 60km are too coarse to represent rainfall patterns well at regional to local scales across the UK and would need to be downscaled before being used in drought assessments, alongside the MaRIUS Weather@Home dataset (Dobson et al., 2020).

Table 1. Evaluation of climate data products against core principles outlined in Section 2. The symbols demonstrate whether the product meets all ('++'), many ('+'), some ('-') or none ('--') of the requirements for drought assessment. A '?' means that we do not know whether this dataset meets the requirement.

DATA PRODUCT	LONG TERM VIEW UNDER CHANGE	BREADTH OF SCENARIOS	HIGH IMPACT LOW LIKELIHOOD EVENTS	SPATIAL COHERENCE AT NATIONAL-SCALE
OBSERVATIONAL DATA	--	--	--	++
UKCP18 PROBABILISTIC PROJECTIONS	++	+	-	--
UKCP18 GLOBAL PROJECTIONS	++	+	?	-
UKCP18 REGIONAL CLIMATE MODEL	++	-	?	++
UKCP18 LOCAL	++	-	?	++
MARIUS WEATHER@HOME	++	-	?	+
STOCHASTICS DATASET (ATKINS)	-	--	+	-
STOCHASTICS DATASET (AME)	-	--	+	-

5. Hydrological and Infrastructure System

There are a wide range of hydrological data and models in use across the UK water sector for planning, management and assessment of hydrological droughts. In this review, we focus on hydrological rainfall-runoff models, hydrological projections from these models and water resource system models as follows:

1. Hydrological models – Grid-to-Grid, DECIPHeR, GR4J, GR6J, CATCHMOD and SHETRAN

2. Water resource system models - WREW, Aquator, Miser, PyWr

3. Hydrological projections – Future Flows, eFLAG, MaRIUS,

We focus on the use of these models for long-term predictions, rather than short-term forecasting. It is worth noting that other tools are available for drought management, such as Qube which the Environment Agency uses to estimate flows in ungauged catchments and the impacts of water use. However, this tool is typically not used for estimating the impacts of climate change on flows which is the focus of this review.

We start this section with a short description of the models and data listed above and then discuss their strengths and weaknesses. The key characteristics of the hydrological and water resource system models and data described in Section 5.1-5.3 are summarised in the appendix in tables A3 and A4.

5.1. Hydrological models for drought management and planning

Hydrological models simulate the movement, distribution, and retention of water within a catchment. These models are typically driven by rainfall and potential evapotranspiration data. They simplify complex physical processes occurring in a catchment using parameters applicable at a wide range of scales. These parameters are determined by using geophysical data such as soils, geology and landcover data and/or by calibrating to observed flow timeseries. Hydrological model outputs are typically river flows but can also include soil moisture and groundwater levels. Typically, these models focus on representing geophysical processes (such as infiltration or runoff generation) while ignoring the effects of human interventions, such as abstractions or reservoir operations, on river flows.

Grid-to-Grid (Bell et al., 2009) is a spatially distributed hydrological model designed to provide a grid-based approach to hydrological modelling. The model is applied nationally across the UK on a 1km grid and requires daily precipitation and potential evapotranspiration as input. The model contains five parameters which are derived from spatial datasets (such as land-cover, soils, geology datasets) and can contain a snow module. The outputs, i.e. soil moisture and flow timeseries, from Grid-to-Grid have been used to analyse past and future river flow and soil moisture droughts, and low flows (Lane and Kay, 2021; Kay et al., 2022). They are also used as part of the UK Hydrological Outlook that provides insights into future hydrological conditions for river flows. The model code is not openly available.

DECIPHeR (Coxon et al., 2019) is a flexible hydrological modelling framework based on Dynamic TOPMODEL (Beven and Freer, 2001) that is designed to be computationally efficient and adaptable to to specific hydrologic settings. DECIPHeR is calibrated nationally across Great Britain and requires daily precipitation and potential evapotranspiration as input. The model contains 7 parameters that are either calibrated using random sampling to observed flow timeseries (Coxon et al., 2019) or through multi-scale parameter regionalisation (Lane et al., 2021) where parameters are derived from spatial datasets (such as land-cover, soils, geology datasets). The outputs, i.e. river flow timeseries have been used to better understand the spatial dynamics of future droughts (Dobson et al., 2020).

Future hydrological projections produced by DECIPHeR are also used by the Environment Agency and Ofwat to inform the implementation of new water resource solutions including water transfers and reservoirs (Ofwat, 2022). The model code is open source and available on github: https://github.com/uob-hydrology/DECIPHeR.

The GR models (Coron et al., 2017) are a suite of conceptual lumped hydrological models designed as simple, quick and easy to run. They require daily precipitation and potential evapotranspiration as input and contain between 4 (GR4J) to 6 (GR6J) free parameters that are typically calibrated to observed flow timeseries. GR4J has been calibrated for many catchments across the UK and the outputs from the model of river flow timeseries have been used to reconstruct historic droughts across Great Britain (Smith et al., 2019), and to evaluate and improve seasonal forecasting of drought (Harrigan et al., 2018). GR6J is commonly used as part of water company water resource management plans (e.g. Anglian Water, 2022) and for supply forecasting and stress testing on extreme droughts (Chan et al., 2022). The model code is open source and available from CRAN: https://cran.r- project.org/web/packages/airGR/index.html.

CATCHMOD (Wilby et al., 1994) is a semi-distributed rainfall-runoff model developed by the Environment Agency. CATCHMOD requires daily precipitation and potential evaporation as input and contains 5 free parameters per response unit that are typically calibrated to observed flow timeseries. It has been used extensively for water resources assessment and climate change impact studies in the Thames catchment in published academic research (Wilby, 2005; Manning et al., 2009) and by the Environment Agency. It is included as part of the Kestrel – IHM software developed by HR Wallingford which enables semi-distributed applications of CATCHMOD and PDM, and has been used for the extreme droughts project. The model code for CATCHMOD is available as an open source python package: https://pypi.org/project/pycatchmod/.

SHETRAN (Lewis et al., 2018b) is a physically based, spatially distributed hydrological model, which has its origins in the Système Hydrologique Europeén (SHE) model (Abbott et al., 1986). SHETRAN requires daily precipitation and potential evapotranspiration as input and a number of spatial geophysical datasets (land cover, elevation geology etc.) for parameterisation. The model has been applied nationally across Great Britain and the outputs (river flow projections) are being used as part of the OpenCLIM project to analyse changes in future floods and droughts using UKCP18 regional projections. The model code is not open source but available on request.

5.2. Water resource system models for drought management and planning

Water resource system (WRS) models simulate the movement of water across a set of interconnected infrastructure for water abstraction, storage, treatment, and distribution. Unlike hydrological models, all WRS models essentially rely on the same conceptualisation, where the water resource system is represented as a network (or graph) of source nodes (e.g. river abstraction points, boreholes, reservoirs), demand nodes (e.g. water treatment works) and links (e.g. rivers, canals and pipes). Their inputs are hydrological inflows (typically surface flows such as streamflow and runoff), water demands and minimum environmental flows. Water demands and hydrological inflows can be derived from either observed or modelled time series. WRS model outputs are flows from/to source/demand nodes, storage levels in reservoirs (calculated by closing water mass balance) and groundwater (calculated by simple empirical relationships between abstractions and levels), energy use and operating costs. WRS models also include water allocation rules to define the spatial distribution of water fluxes across links, and operating rules to define the temporal distribution of water fluxes at storage nodes. Operating rules are typically defined by the user, while water allocation rules can be defined by the user or, most commonly, are calculated at each time step within the model through a linear programming algorithm.

Operating rules can, to some extent, account for outage and water quality issues – which in drought conditions may become very important as deterioration of water quality means that treatment works capacity may become the limiting factor to abstractions - although they do so in a very simplified way. While conceptually very similar to one another, the key differences between WRS models is in their software implementation, i.e. runtime (slow vs fast), accessibility of the source code (proprietary vs open-source), and type of interface (command line vs Graphical User Interface, GUI).

Aquator (Hydro International, 2023) and MISER (Fowler et al., 1999) are proprietary software packages widely used by UK water companies to simulate WRSs at water resource zone level. They come with a sophisticated GUI that facilitates building and running the model. GUIs can also handle repeated model simulations, such as optimisation of operating rules parameters or simulation of hundreds of scenarios, with modifications to permit batch processing.

Pywr (Tomlinson et al., 2020) is a freely available open-source, command-line based Python package. It is used by individual water companies as well as for national-scale studies such as analysing the potential of different water transfers combinations for satisfying future water needs (Sec. 9.3.3 of the National Framework for water resources (EA, 2020)).

WREW (Water Resources England and Wales) is a customised version of the WATHNET software (Kuczera, 1992) specifically developed by Dobson et al (2020) for national-scale applications. It reproduces all major water supply infrastructure in England and Wales that are connected into wider water network via any river or transfer of significance (i.e., >2 Ml/day). Similar to Pywr, WREW is computationally fast, however it is not openly available.

5.3. Hydrological projections for drought management and planning

The Future Flows hydrology dataset provides hydrological projections for the UK based on an 11-member ensemble of transient projections that forms part of UKCP09 (Prudhomme et al., 2013). It includes projections from multiple hydrological models (CERF, PDM, CLASSIC) and one groundwater model (R-Groundwater). Daily river flow projections are provided for 1951-2098 for 281 catchments. Monthly groundwater levels are provided for 24 boreholes. These projections are openly available on the Environmental Information Data Centre (Haxton, 2012). While this dataset was commonly used across the UK water sector, they are based on the older set of UK Climate Projections and thus are now less commonly used for academic, industry and regulatory drought assessments.

eFLAG (enhanced future FLows and Groundwater') dataset provides hydrological projections for the UK based on the UKCP18 regional projections (Hannaford et al., 2022). It includes projections from multiple hydrological models (Grid-to-Grid, PDM, GR4J and GR6J) and groundwater models for groundwater levels (Aquimod) and recharge (ZOODRM).

Projections are provided for 200 catchments, 54 groundwater level boreholes and 558 groundwater bodies from 1981-2080. These projections are openly available on the Environmental Information Data Centre (Hannaford, 2022).

The MARIUS hydrological projections provide two sets of hydrological projections. The first set of MARIUS hydrological projections are for England and Wales and based on the MARIUS weather@Home climate projections and DECIPHeR hydrological modelling framework. It includes timeseries up to 2100 for 338 locations across England and Wales. Some of these locations are on river gauges, while others are strategically important points for water resource system models. These future river flow timeseries were used as input into the water resource system model, WATHNET, to analyse future spatial dynamics of droughts (Dobson et al., 2020) and are now used by the Environment Agency and Ofwat to test water resource system options. The projections are openly available on the Bristol research data repository at https://data.bris.ac.uk/data/dataset/2pkv9oxgfzvts235zrui7xz00g. The second set of MARIUS hydrological projections are for Great Britain using the Grid-to- Grid hydrological model (Bell et al., 2018). This provides a 100-member ensemble of monthly mean flow and soil moisture on a 1km grid for three timeslices up to 2100. The projections are openly available on the EIDC at https://catalogue.ceh.ac.uk/documents/3b90962e-6fc8-4251-853e-b9683e37f790.

5.4. Strengths and weaknesses of current hydrological and water resource system models and projections

The hydrological models and datasets described above meet different needs for drought management, assessment, and planning, and thus have varying strengths and weaknesses.

All the hydrological and water resource system models and projections meet the need for ensuring a **long view under change** in drought assessments. The models are all sufficiently computationally efficient to run long time-series and have been used to produce projections near to the end of the 21st century.

While some of the models enable a **breadth of scenarios** to be considered, others are expensive to run and would require a lot of time and computing power to run thousands of scenarios for long timeseries. In particular, the gridded hydrological models (such as Grid-to- Grid and SHETRAN) are more expensive to run compared to lumped conceptual hydrological models (such as GR4J or GR6J). Even when hydrological models can in principle be run against a range of climate scenarios, the breadth of hydrological projections generated is limited by the breadth of available climate scenarios. At present, this is extremely narrow as all the UK hydrological projections use climate projections from a single climate model and emission scenario (RCP8.5 for eFLAG and MaRIUS, SRES A1B for Future Flows).

While some of the hydrological models include representation of the effects of human activities on hydrological processes (for instance Rameshwaran et al 2022 explicitly included river abstractions into G2G and CATCHMOD includes abstractions and discharges), they still mainly focus on natural processes. Hence the predictions derived from these models do not (and at present, can not) consider the long-term effects of changes in society (e.g. changes to environmental regulations) or the economic (e.g. changes in demand). In contrast, water resource models can easily consider changes in future water demand, future abstraction licences and/or environmental flows as these are all explicitly considered as inputs to these models. However, the scenarios of how these inputs may evolve over long temporal scales are highly uncertain, possibly even more than climate projections (Walker, 2013). Thus, one key practical consideration is ensuring water resource system models can be run against a large breadth of socio economic scenarios. This may prove difficult with WRS models that are not run from the command line or that do not have batch processing built into the GUI, as manually inputting a large number of inputs scenarios is extremely time consuming and may be practically unfeasible.

For the assessment of **high impact, low likelihood extreme droughts,** all the hydrological models have the capability of assessing these types of events given appropriate input data. A key consideration for extremes is, however, appropriate model calibration and the capacity of the models to handle abrupt, non-linear changes in rainfall-runoff responses following severe droughts. Previous studies have demonstrated that many hydrological models poorly capture hydrological flows during extreme, multi-annual droughts (Fowler et al., 2016). For example, Fowler et al (2022) show how commonly used hydrological models (in this case GR4J but the result applies to many models) poorly represent shifts in hydrological behaviour during multi-annual droughts and can't be relied upon to anticipate future changes. These challenges arise due to the choice of historic period chosen for calibration and the fact that are evaluation criteria to calibrate model parameters, such as NSE, logNSE or RMSE (used in eFLAG and MaRIUS hydrological projections) aim at capturing a good average model performance, but not necessarily result in good assessment of model performance during extreme events. Consequently, model output may under or over- estimate future droughts. While the hydrological projections discussed in this review have been used to evaluate the impact of climate change on future droughts, there has been less focus on extreme multi-annual droughts.

For WRS models, one key challenge in simulating high impact, low likelihood extreme droughts is that the operating rules implemented in these models typically reproduce operations in normal conditions. During extreme drought events, these rules do not apply as system managers will often respond to very extreme events by taking special measures, beyond their standard drought management strategies, possibly through negotiations with stakeholders and regulators (e.g. EA). This informal decision making process is impossible to code into a simulation model and thus WRS models tend to perform poorly during extreme events (e.g. Rougé et al., 2021) for an application to a large water system in the US).

Finally, while many of the distributed hydrological models (G2G and DECIPHeR) enable **spatially coherent national-scale** hydrological projections, the outputs are currently limited and model performance is variable at a national-scale.

The eFLAG projections contain future projections for 200 catchments across Great Britain, while the MaRIUS hydrological projections contains future projections for 338 locations across England and Wales. These locations do not necessarily coincide with the inflow points needed by water resource system models or other impacts models. Lumped catchment models such as GR4J/GR6J can easily be run at national scale but don't offer the same spatially consistent parameter fields. WREW covers 70 water resource zones and 16 water utility companies across England and Wales, and Pywr can be set up to run at national scale (as done in EA (2020)). A key issue here though is that many of the data needed to set-up and calibrate such water resources models are propriety of water companies and not openly available. Another challenge with national-scale hydrological projections is ensuring 'good' model performance everywhere. Current national-scale hydrological models often perform poorly in groundwater-dominated, drier and heavily human-influenced regions of the UK (Coxon et al., 2019). Projections may be less robust in these regions as a result.

6. Conclusions and Recommendations

In summary, this report has reviewed a wide range of data and models in use across the UK water sector that demonstrates the progression in this sector over the past decades. This review highlights our strengths in producing long term, spatially coherent, national-scale climate, hydrological and water resource projections for drought management, planning and assessment. It also highlights the wide range of data and models available, which represents an exciting opportunity to learn from the multitude of approaches. However, it has also shown that our current suite of models and data all suffer from similar deficiencies. In particular, we lack a wide range of climate and socio-economic scenarios, and we lack the ability to assess high impact, low likelihood extreme droughts. Based on the critique above, we provide three key recommendations for models and data in use within the water sector.

6.1. We need more structured approaches to prioritise efforts for improvement of datasets and models

As this review shows, the datasets and models available to assist droughts planning all have different strengths and limitations. This means that the directions that could be explored to improve the state-of-art are manifold, and it becomes essential to identify priorities in a coherent way. Efforts for data and model improvements should be informed by a better understanding of key sensitivities of drought drivers and metrics. For example, precipitation may be the dominant driver for hydrological droughts in certain catchments, whereas in others the dominant driver could be the groundwater contribution to streamflow, implying that representing groundwater processes correctly is the top priority. Or, uncertainties in the supply side of the drought assessment may be unimportant relative to the uncertainty brought in by the estimation of future demand.

Indeed, previous sensitivity analysis studies have shown that aggregated output metrics of complex environmental models are typically controlled by a relatively small number of model input factors (Wagener and Pianosi, 2019). Identifying such dominant controls in the modelling process can help prioritise efforts for reducing uncertainty where most effective and avoid investments in improving dataset/model components that would have limited effects on the droughts assessment results. This aligns with the need to complement top-down approaches (like those evaluated in this review) with bottom-up approaches where sensitivity testing of adaptation options take centre stage in drought management (Wilby and Dessai, 2010).

6.2. We need more structured approaches to document, evaluate and compare existing and future datasets and models

Another message emerging from this review is the substantial number of datasets and models for droughts planning at present that will likely continue to increase in the future, as this remains a very active area of research and development. To support users in navigating this increasingly complex landscape, it is important to make appropriate choices of which model/dataset to use. For example, models should be evaluated and compared using a more structured approach to document models, and particularly their underlying assumptions and scope of validity (Wagener et al., 2021). This will require sharing data and model (code), but also experimental procedures (i.e. modelling protocols) to ensure the comparability and reproducibility of simulation experiments (Ceola et al., 2015), and potentially establishing ISO-like 'standard' for drought assessment (see for example Climate Sense, (2022) for climate services)

6.3. We need to develop greater breadth of scenarios and modelling frameworks to assess them

One identified gap from this review is the limited range of scenarios available for drought management and planning. Climate scenarios are currently limited to a single emission scenario and climate model. The new EUROCORDEX-UK project will extend the UKCP18 suite of climate projections by including a more comprehensive sampling of modelling uncertainty (see https://www.ucl.ac.uk/statistics/research/eurocordex-uk). There are also exciting developments in generating new projections of high impact, low likelihood events. The new NERC funded Climate change in the Arctic-North Atlantic region and impact on the UK (CANARI) project will focus on producing storylines of extreme weather. These storylines could be used to better understand system vulnerabilities to droughts and the extent to which future climate risks can be managed by specific adaptation policies (Shepherd et al., 2018).

However, considering the climate scenarios alone would be a poor predictor of future hydrological droughts because of the many complex ways humans use and alter terrestrial water fluxes to cope with climate variability. Hydrological models typically assess climate change impacts on hydrological droughts (e.g. in terms of low flows) while water resource system models assess socio-economic-droughts (e.g. in terms of reduced supply). This one- way flow of data from hydrological models into water resource system models could miss important two-way interactions and feedbacks between terrestrial water systems and people. Increasingly, more integrated modelling frameworks are being developed where abstractions, discharges and reservoir systems are being included into hydrological models to capture these interactions. The increased feedback of social water systems and hydrological systems is valuable, although it is important to consider the represented feedback(s) and the temporal and/or spatial scale of hydrological and social components in these integrated models (Vanelli et al., 2022). For example, integrated models (or socio- hydrological models) may represent a singular water demand/supply response related to a policy intervention, but they could also represent an interactive policy or agent-based response that can capture adaptation of societal water demand (Garcia et al., 2016; Wens et al., 2019). New future drought projections including changes in water demand are forthcoming through the Climate Services for a Net Zero Resilient World (CS-NOW).

However, more research will be needed to think about how we best incorporate human-water interactions in hydrological models, particularly for data that are currently not widely available (e.g. abstractions data, reservoir operating rules).

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References

Abbott, M. B., Bathurst, J. C., Cunge, J. A., O'Connell, P. E., and Rasmussen, J.: An introduction to the European Hydrological System — Systeme Hydrologique Europeen, "SHE", 1: History and philosophy of a physically-based, distributed modelling system, Journal of Hydrology, 87, 45–59, https://doi.org/10.1016/0022-1694(86)90114-9, 1986.

Environment Agency Potential Evapotranspiration Dataset: https://www.data.gov.uk/dataset/7b58506c-620d-433c-afce-d5d93ef7e01e/environment-agency-potential-evapotranspiration-dataset, last access: 9 January 2023.

Anglian Water: Draft WRMP24 Technical Document : Supply Forecast, 2022.

Arnell, N. W.: Uncertainty in the relationship between climate forcing and hydrological response in UK catchments, Hydrol. Earth Syst. Sci., 15, 897–912, https://doi.org/10.5194/hess-15-897-2011, 2011.

Atkins: Regional Climate Data Tools, 2020.

Bell, V. A., Kay, A. L., Jones, R. G., Moore, R. J., and Reynard, N. S.: Use of soil data in a grid-based hydrological model to estimate spatial variation in changing flood risk across the UK, Journal of Hydrology, 377, 335–350, https://doi.org/10.1016/j.jhydrol.2009.08.031, 2009.

Bell, V. A., Kay, A. L., Rudd, A. C., and Davies, H. N.: The MaRIUS-G2G datasets: Grid-to- Grid model estimates of flow and soil moisture for Great Britain using observed and climate model driving data, Geoscience Data Journal, 5, 63–72, https://doi.org/10.1002/gdj3.55, 2018.

Beven, K. and Freer, J.: A dynamic TOPMODEL, Hydrol. Process., 15, 1993–2011, https://doi.org/10.1002/hyp.252, 2001.

Brown, E. M., M. J. ;Robinson, E. L. ;Kay, A. L. ;Chapman, R. ;Bell, V. A. ;Blyth: Potential evapotranspiration derived from HadUK-Grid 1km gridded climate observations 1969-2021 (Hydro-PE HadUK-Grid), , https://doi.org/10.5285/9275ab7e-6e93-42bc-8e72-59c98d409deb, 2022.

Byers, E. A., Coxon, G., Freer, J., and Hall, J. W.: Drought and climate change impacts on cooling water shortages and electricity prices in Great Britain, Nat Commun, 11, 2239, https://doi.org/10.1038/s41467-020-16012-2, 2020.

Ceola, S., Arheimer, B., Baratti, E., Blöschl, G., Capell, R., Castellarin, A., Freer, J., Han, D., Hrachowitz, M., Hundecha, Y., Hutton, C., Lindström, G., Montanari, A., Nijzink, R., Parajka, J., Toth, E., Viglione, A., and Wagener, T.: Virtual laboratories: new opportunities for collaborative water science, Hydrol. Earth Syst. Sci., 19, 2101–2117, https://doi.org/10.5194/hess-19-2101-2015, 2015.

Chan, W. C. H., Shepherd, T. G., Facer-Childs, K., Darch, G., and Arnell, N. W.: Storylines of UK drought based on the 2010–2012 event, Hydrology and Earth System Sciences, 26, 1755–1777, https://doi.org/10.5194/hess-26-1755-2022, 2022.

Chengot, R., Knox, J. W., Coxon, G., Cojocaru, G., and Holman, I. P.: An enhanced version of the D-Risk decision support webtool for multi-scale management of water abstraction and drought risks in irrigated agriculture, Computers and Electronics in Agriculture, 204, 107516, https://doi.org/10.1016/j.compag.2022.107516, 2023.

Christierson, B. V., Vidal, J.-P., and Wade, S. D.: Using UKCP09 probabilistic climate information for UK water resource planning, Journal of Hydrology, 424–425, 48–67, https://doi.org/10.1016/j.jhydrol.2011.12.020, 2012.

Climate Sense: Climate Services: Principles, requirements, and guidelines, 2022.

Coron, L., Thirel, G., Delaigue, O., Perrin, C., and Andréassian, V.: The suite of lumped GR hydrological models in an R package, Environmental Modelling & Software, 94, 166–171, https://doi.org/10.1016/j.envsoft.2017.05.002, 2017.

Coxon, G., Freer, J., Lane, R., Dunne, T., Knoben, W. J. M., Howden, N. J. K., Quinn, N., Wagener, T., and Woods, R.: DECIPHeR v1: Dynamic fluxEs and ConnectIvity for Predictions of HydRology, Geoscientific Model Development, 12, 2285–2306, https://doi.org/10.5194/gmd-12-2285-2019, 2019.

Coxon, G., Addor, N., Bloomfield, J. P., Freer, J., Fry, M., Hannaford, J., Howden, N. J. K., Lane, R., Lewis, M., Robinson, E. L., Wagener, T., and Woods, R.: CAMELS-GB: hydrometeorological time series and landscape attributes for 671 catchments in Great Britain, Earth System Science Data, 12, 2459–2483, https://doi.org/10.5194/essd-12-2459- 2020, 2020.

Coxon, R., G. ;Addor, N. ;Bloomfield, J. P. ;Freer, J. ;Fry, M. ;Hannaford, J. ;Howden, N. J.K. ;Lane, R. ;Lewis, M. ;Robinson, E. L. ;Wagener, T. ;Woods: Catchment attributes and hydro-meteorological timeseries for 671 catchments across Great Britain (CAMELS-GB), , https://doi.org/10.5285/8344e4f3-d2ea-44f5-8afa-86d2987543a9, 2020.

Dawkins, L. C., Osborne, J. M., Economou, T., Darch, G. J. C., and Stoner, O. R.: The Advanced Meteorology Explorer: a novel stochastic, gridded daily rainfall generator, Journal of Hydrology, 607, 127478, https://doi.org/10.1016/j.jhydrol.2022.127478, 2022.

Dobson, B., Coxon, G., Freer, J., Gavin, H., Mortazavi-Naeini, M., and Hall, J. W.: The Spatial Dynamics of Droughts and Water Scarcity in England and Wales, Water Resources Research, 56, e2020WR027187, https://doi.org/10.1029/2020WR027187, 2020.

Environment Agency: Meeting our future water needs: a national framework for water resources, 91, 2020.

Environment Agency: Water resources planning guideline, 2022.

Fowler, K. J. A., Peel, M. C., Western, A. W., Zhang, L., and Peterson, T. J.: Simulating runoff under changing climatic conditions: Revisiting an apparent deficiency of conceptual rainfall-runoff models, Water Resources Research, 52, 1820–1846, https://doi.org/10.1002/2015WR018068, 2016.

Fowler, M., Cook, S., and Lumbers, J.: Practical experience in the successful implementation of optimisation systems for water supply management, Computing and Control for the Water Industry, 1999.

Garcia, M., Portney, K., and Islam, S.: A question driven socio-hydrological modeling process, Hydrology and Earth System Sciences, 20, 73–92, https://doi.org/10.5194/hess-20- 73-2016, 2016.

Guillod, B. P., Jones, R. G., Dadson, S. J., Coxon, G., Bussi, G., Freer, J., Kay, A. L., Massey, N. R., Sparrow, S. N., Wallom, D. C. H., Allen, M. R., and Hall, J. W.: A large set of potential past, present and future hydro-meteorological time series for the UK, Hydrol. Earth Syst. Sci. Discuss., 2017, 1–39, https://doi.org/10.5194/hess-2017-246, 2017.

Hannaford, A., J. ;Mackay, J. ;Ascot, M. ;Bell, V. ;Chitson, T. ;Cole, S. ;Counsell, C. ;Durant,M. ;Facer-Childs, K. ;Jackson, C. ;Kay, A. ;Lane, R. ;Mansour, M. ;Moore, R. J. ;Parry,S. ;Rudd, A. ;Simpson, M. ;Turner, S. ;Wallbank, J. ;Wells, S. ;Wilcox: Hydrological projections for the UK, based on UK Climate Projections 2018 (UKCP18) data, from the Enhanced Future Flows and Groundwater (eFLaG) project, , https://doi.org/10.5285/1bb90673-ad37-4679-90b9-0126109639a9, 2022.

Hannaford, J., Mackay, J., Ascott, M., Bell, V., Chitson, T., Cole, S., Counsell, C., Durant, M., Jackson, C. R., Kay, A., Lane, R., Mansour, M., Moore, R., Parry, S., Rudd, A., Simpson, M., Facer-Childs, K., Turner, S., Wallbank, J., Wells, S., and Wilcox, A.: eFLaG: enhanced future FLows and Groundwater. A national dataset of hydrological projections based on UKCP18, Earth System Science Data Discussions, 1–40, https://doi.org/10.5194/essd-2022- 40, 2022.

Harrigan, S., Prudhomme, C., Parry, S., Smith, K., and Tanguy, M.: Benchmarking ensemble streamflow prediction skill in the UK, Hydrology and Earth System Sciences, 22, 2023–2039, https://doi.org/10.5194/hess-22-2023-2018, 2018.

Haxton, C., T. ;Crooks, S. ;Jackson, C. R. ;Barkwith, A. K. A. P. ;Kelvin, J. ;Williamson, J. ;Mackay, J. D. ;Wang, L. ;Davies, H. ;Young, A. ;Prudhomme: Future flows hydrology data,, https://doi.org/10.5285/f3723162-4fed-4d9d-92c6-dd17412fa37b, 2012.

Hollis, D., McCarthy, M., Kendon, M., Legg, T., and Simpson, I.: HadUK-Grid—A new UK dataset of gridded climate observations, Geosci Data J, 6, 151–159, https://doi.org/10.1002/gdj3.78, 2019.

HR Wallingford: CCRA2: Updated projections for water availability for the UK, Final Report - Appendices, 2015.

Hydro-Logic Aquator XV User Guide Version 5, Hydro international, Clevedon, UK, 2023.

Kay, A. L., Griffin, A., Rudd, A. C., Chapman, R. M., Bell, V. A., and Arnell, N. W.: Climate change effects on indicators of high and low river flow across Great Britain, Advances in Water Resources, 151, 103909, https://doi.org/10.1016/j.advwatres.2021.103909, 2021.

Kay, A. L., Lane, R. A., and Bell, V. A.: Grid-based simulation of soil moisture in the UK: future changes in extremes and wetting and drying dates, Environ. Res. Lett., 17, 074029, https://doi.org/10.1088/1748-9326/ac7a4e, 2022.

Keller, V. D. J., Tanguy, M., Prosdocimi, I., Terry, J. A., Hitt, O., Cole, S. J., Fry, M., Morris, D. G., and Dixon, H.: CEH-GEAR: 1 km resolution daily and monthly areal rainfall estimates for the UK for hydrological and other applications, Earth System Science Data, 7, 143–155, https://doi.org/10.5194/essd-7-143-2015, 2015.

Kuczera, G.: Water supply headworks simulation using network linear programming, Advances in Engineering Software, 14, 55–60, https://doi.org/10.1016/0965-9978(92)90084- S, 1992.

Lane, R. A. and Kay, A. L.: Climate Change Impact on the Magnitude and Timing of Hydrological Extremes Across Great Britain, Frontiers in Water, 3, 2021.

Lane, R. A., Freer, J. E., Coxon, G., and Wagener, T.: Incorporating Uncertainty Into Multiscale Parameter Regionalization to Evaluate the Performance of Nationally Consistent Parameter Fields for a Hydrological Model, Water Resources Research, 57, e2020WR028393, https://doi.org/10.1029/2020WR028393, 2021.

Lane, R. A., Coxon, G., Freer, J., Seibert, J., and Wagener, T.: A large-sample investigation into uncertain climate change impacts on high flows across Great Britain, Hydrology and Earth System Sciences, 26, 5535–5554, https://doi.org/10.5194/hess-26-5535-2022, 2022.

Lewis, E., Quinn, N., Blenkinsop, S., Fowler, H. J., Freer, J., Tanguy, M., Hitt, O., Coxon, G., Bates, P., and Woods, R.: A rule based quality control method for hourly rainfall data and a 1 km resolution gridded hourly rainfall dataset for Great Britain: CEH-GEAR1hr, Journal of Hydrology, 564, 930–943, https://doi.org/10.1016/j.jhydrol.2018.07.034, 2018a.

Lewis, E., Birkinshaw, S., Kilsby, C., and Fowler, H. J.: Development of a system for automated setup of a physically-based, spatially-distributed hydrological model for catchments in Great Britain, Environmental Modelling & Software, 108, 102–110, https://doi.org/10.1016/j.envsoft.2018.07.006, 2018b.

Lewis, E., Quinn, N., Blenkinsop, S., Fowler, H. J., Freer, J., Tanguy, M., Hitt, O., Coxon, G., Bates, P., Woods, R., Fry, M., Chevuturi, A., Swain, O., and White, S. M.: Gridded estimates of hourly areal rainfall for Great Britain 1990-2016 [CEH-GEAR1hr] v2, 2022.

Lowe, J. A., Bernie, D., Bett, P., Bricheno, L., Brown, S., Calvert, D., Clark, R., Edwards, T., Fosser, G., Fung, F., Gohar, L., Good, P., Gregory, J., Harris, G., Howard, T., Kaye, N., Kendon, E., Krijnen, J., Maisey, P., McDonald, R., McInnes, R., McSweeney, C., Mitchell, J. F. B., Murphy, J., Palmer, M., Roberts, C., Rostron, J., Thornton, H., Tinker, J., Tucker, S., Yamazaki, K., and Belcher, S.: UKCP18 Science Overview Report, 73, 2018.

Manning, L. J., Hall, J. W., Fowler, H. J., Kilsby, C. G., and Tebaldi, C.: Using probabilistic climate change information from a multimodel ensemble for water resources assessment, Water Resources Research, 45, https://doi.org/10.1029/2007WR006674, 2009.

Met Office, Hollis, D., McCarthy, M., Kendon, M., and Legg, T.: HadUK-Grid Gridded Climate Observations on a 1km grid over the UK, v1.1.0.0 (1836-2021), https://doi.org/10.5285/BBCA3267DC7D4219AF484976734C9527, 2022.

Murgatroyd, A., Gavin, H., Becher, O., Coxon, G., Hunt, D., Fallon, E., Wilson, J., Cuceloglu, G., and Hall, J. W.: Strategic analysis of the drought resilience of water supply systems, Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences, 380, 20210292, https://doi.org/10.1098/rsta.2021.0292, 2022.

Murphy, J. M., Harris, G. R., Sexton, D. M. H., Kendon, E. J., Bett, P. E., Clark, R. T., Eagle, K. E., Fosser, G., Fung, F., Lowe, J. A., McDonald, R. E., McInnes, R. N., McSweeney, C. F., Mitchell, J. F. B., Rostron, J. W., Thornton, H. E., Tucker, S., and Yamazaki, K.: UKCP18 Land Projections: Science Report, 191, 2018.

National Infrastructure Commission: Preparing for a drier future, 31, 2018.

Ofwat: The National System Simulation Modelling (NSSM) Project - Phase 2 Report - July 2022, 2022.

Prudhomme, C., Haxton, T., Crooks, S., Jackson, C., Barkwith, A., Williamson, J., Kelvin, J., Mackay, J., Wang, L., Young, A., and Watts, G.: Future Flows Hydrology: an ensemble of daily river flow and monthly groundwater levels for use for climate change impact assessment across Great Britain, Earth System Science Data, 5, 101–107, https://doi.org/10.5194/essd-5-101-2013, 2013.

Robinson, A. C., E. L. ;Blyth, E. M. ;Clark, D. B. ;Comyn-Platt, E. ;Rudd: Climate hydrology and ecology research support system potential evapotranspiration dataset for Great Britain (1961-2017) [CHESS-PE], , https://doi.org/10.5285/9116e565-2c0a-455b-9c68- 558fdd9179ad, 2020.

Rougé, C., Reed, P. M., Grogan, D. S., Zuidema, S., Prusevich, A., Glidden, S., Lamontagne, J. R., and Lammers, R. B.: Coordination and control – limits in standard representations of multi-reservoir operations in hydrological modeling, Hydrology and Earth System Sciences, 25, 1365–1388, https://doi.org/10.5194/hess-25-1365-2021, 2021.

Serinaldi, F. and Kilsby, C. G.: A modular class of multisite monthly rainfall generators for water resource management and impact studies, Journal of Hydrology, 464–465, 528–540, https://doi.org/10.1016/j.jhydrol.2012.07.043, 2012.

Shepherd, T. G., Boyd, E., Calel, R. A., Chapman, S. C., Dessai, S., Dima-West, I. M., Fowler, H. J., James, R., Maraun, D., Martius, O., Senior, C. A., Sobel, A. H., Stainforth, D. A., Tett, S. F. B., Trenberth, K. E., van den Hurk, B. J. J. M., Watkins, N. W., Wilby, R. L., and Zenghelis, D. A.: Storylines: an alternative approach to representing uncertainty in physical aspects of climate change, Climatic Change, 151, 555–571, https://doi.org/10.1007/s10584-018-2317-9, 2018.

Smith, K. A., Barker, L. J., Tanguy, M., Parry, S., Harrigan, S., Legg, T. P., Prudhomme, C., and Hannaford, J.: A multi-objective ensemble approach to hydrological modelling in the UK: an application to historic drought reconstruction, Hydrology and Earth System Sciences, 23, 3247–3268, https://doi.org/10.5194/hess-23-3247-2019, 2019.

Tallaksen, L. and Lanen, H. A. J. van: Hydrological drought. Processes and estimation methods for streamflow and groundwater, Elsevier, 2004.

Tanguy, J., M. ;Prudhomme, C. ;Smith, K. ;Hannaford: Historic Gridded Potential Evapotranspiration (PET) based on temperature-based equation McGuinness-Bordne calibrated for the UK (1891-2015), , https://doi.org/10.5285/17b9c4f7-1c30-4b6f-b2fe-f7780159939c, 2017.

Tanguy, V. D. J., M. ;Dixon, H. ;Prosdocimi, I. ;Morris, D. G. ;Keller: Gridded estimates of daily and monthly areal rainfall for the United Kingdom (1890-2019) [CEH-GEAR], , https://doi.org/10.5285/dbf13dd5-90cd-457a-a986-f2f9dd97e93c, 2021.

Thames Water: Draft WRMP24 – Section 4: Current and Future Water Supply, 2022.

Tomlinson, J. E., Arnott, J. H., and Harou, J. J.: A water resource simulator in Python, Environmental Modelling & Software, 126, 104635, https://doi.org/10.1016/j.envsoft.2020.104635, 2020.

Van Loon, A. F.: Hydrological drought explained, WIREs Water, 2, 359–392, https://doi.org/10.1002/wat2.1085, 2015.

Van Loon, A. F., Stahl, K., Di Baldassarre, G., Clark, J., Rangecroft, S., Wanders, N., Gleeson, T., Van Dijk, A. I. J. M., Tallaksen, L. M., Hannaford, J., Uijlenhoet, R., Teuling, A. J., Hannah, D. M., Sheffield, J., Svoboda, M., Verbeiren, B., Wagener, T., and Van Lanen,

H. A. J.: Drought in a human-modified world: reframing drought definitions, understanding, and analysis approaches, Hydrology and Earth System Sciences, 20, 3631–3650, https://doi.org/10.5194/hess-20-3631-2016, 2016.

Vanelli, F. M., Kobiyama, M., and de Brito, M. M.: To which extent are socio-hydrology studies truly integrative? The case of natural hazards and disaster research, Hydrology and Earth System Sciences, 26, 2301–2317, https://doi.org/10.5194/hess-26-2301-2022, 2022.

Wagener, T. and Pianosi, F.: What has Global Sensitivity Analysis ever done for us? A systematic review to support scientific advancement and to inform policy-making in earth system modelling, Earth-Science Reviews, 194, 1–18. https://doi.org/10.1016/j.earscirev.2019.04.006, 2019.

Wagener, T., Gleeson, T., Coxon, G., Hartmann, A., Howden, N., Pianosi, F., Rahman, M., Rosolem, R., Stein, L., and Woods, R.: On doing hydrology with dragons: Realizing the value of perceptual models and knowledge accumulation, WIREs Water, 8, e1550, https://doi.org/10.1002/wat2.1550, 2021.

Walker, G.: A critical examination of models and projections of demand in water utility resource planning in England and Wales, International Journal of Water Resources Development, 29, 352–372, https://doi.org/10.1080/07900627.2012.721679, 2013.

Watts, G., Battarbee, R. W., Bloomfield, J. P., Crossman, J., Daccache, A., Durance, I., Elliott, J. A., Garner, G., Hannaford, J., Hannah, D. M., Hess, T., Jackson, C. R., Kay, A. L., Kernan, M., Knox, J., Mackay, J., Monteith, D. T., Ormerod, S. J., Rance, J., Stuart, M. E., Wade, A. J., Wade, S. D., Weatherhead, K., Whitehead, P. G., and Wilby, R. L.: Climate change and water in the UK – past changes and future prospects, Progress in Physical Geography: Earth and Environment, 39, 6–28, https://doi.org/10.1177/0309133314542957, 2015.

Wens, M., Johnson, J. M., Zagaria, C., and Veldkamp, T. I. E.: Integrating human behavior dynamics into drought risk assessment—A sociohydrologic, agent-based approach, WIREs Water, 6, e1345, https://doi.org/10.1002/wat2.1345, 2019.

Wessex Water: Water Resource Management Plan 2024 Main Technical Report, 2023.

Wilby, R., Greenfield, B., and Glenny, C.: A coupled synoptic-hydrological model for climate change impact assessment, Journal of Hydrology, 153, 265–290, https://doi.org/10.1016/0022-1694(94)90195-3, 1994.

Wilby, R. L.: Uncertainty in water resource model parameters used for climate change impact assessment, Hydrological Processes, 19, 3201–3219, https://doi.org/10.1002/hyp.5819, 2005.

Wilby, R. L. and Dessai, S.: Robust adaptation to climate change, Weather, 65, 180–185, https://doi.org/10.1002/wea.543, 2010.

Wilby, R. L. and Dessai, S.: Robust adaptation to climate change, Weather, 65, 180–185, https://doi.org/10.1002/wea.543, 2010.

Appendix

Table A1 | Description, key characteristics, and references of observational climate data products

Data Product	Description	Spatial & Temporal Resolution	Spatial/ Temporal Coverage	Uncertainty	Key References
HAD-UK RAINFALL	Inverse distance weighted interpolation of Met Office national database of observed precipitation	1km, daily	United Kingdom, 1891-2022	No uncertainty estimates	(Watts et al., 2015)
CEH-GEAR RAINFALL	Nearest neighbour interpolation of Met Office national database of observed precipitation	1km, daily	United Kingdom, 1890-2019	Data on distance to closest rain gauge	(Tanguy, 2021; Keller et al., 2015)
CEH-GEAR1HR RAINFALL	Nearest neighbour interpolation of a national database of hourly raingauge observations. These are then used to temporally disaggregate the CEH- GEAR daily rainfall.	1km, hourly	Great Britain, 1990-2016	Data on distance to closest rain gauge and quality indicators	(Lewis et al., 2018a, 2022)
HYDRO-PE (FROM HAD-UK GRID)	Calculated from Had-UK meteorological data using the Penman-Monteith equation parameterised for a well-watered grass surface.	1km, daily	United Kingdom, 1969-2021	Includes two PET products – with and without interception correction	(Brown, 2022)
CHESS-PE	Calculated from CHESS-met meteorological data (downscaled from 40km) using the Penman- Monteith equation parameterised for a well-watered grass surface.	1km, daily	Great Britain, 1961-2017	Includes two PET products – with and without interception correction	(Robinson, 2020)
HISTORIC GRIDDED PET	Calculated from temperature data using the McGuinness-Bordne equation.	5km, daily	United Kingdom, 1891-2015	No uncertainty estimates	(Tanguy, 2017)
ENVIRONMENT AGENCY PET	Calculated from MIDAS data and gridded to 1km. Calculated PET using the Penman-Monteith equation parameterised for a well-watered grass	1km, daily	England and Wales, 1961-date	Includes two PET products – with and without interception	(Environment Agency Potential Evapotranspiration

	surface.			correction	Dataset, 2023)
CAMELS-GB	Catchment average timeseries derived from CEH- GEAR and CHESS-PE	Catchment averaged, daily	671 Great Britain catchments, 1970-2015	Includes two PET products – with and without interception correction	(Coxon, 2020; Coxon et al., 2020)

Table A2 | Description, key characteristics, and references of modelled climate data products

Data Product	Spatial/ And Temporal Resolution	Spatial/ Temporal Coverage	Spatially/ Temporally Coherent?	No. Ensemble Members	Future Climate Uncertainty	Bias Correction Applied?	Refs
UKCP18 PROBABILISTIC PROJECTIONS	25km, monthly	UK, 1961-2100	No	3000	Five emission scenarios (RCP2.6, RCP4.5, RCP6.0, RCP8.5,SRESA1B), multiple climate models, perturbed parameter ensemble	No bias correction needed	Murphy et al., 2018
UKCP18 GLOBAL PROJECTIONS	60km, daily	Global, 1900-2100	Yes	28	One emission scenario (RCP8.5), CMIP5 Climate Models and Met Office Global Model, perturbed parameter ensemble	Bias correction required, no bias corrected products available	Murphy et al., 2018
UKCP18 REGIONAL CLIMATE MODEL	12km, daily	UK, 1981-2080	Yes	12	One emission scenario (RCP8.5), one climate model (Met Office Regional Model), perturbed parameter ensemble	Bias correction required, some bias corrected products available	Murphy et al., 2018
UKCP18 LOCAL	2.2km, daily	UK, Three time slices (1981-2000, 2021-2040 and 2061-2080)	Yes	12	One emission scenario (RCP8.5), one climate model (Met Office Regional Model)	Bias correction potentially required, no bias corrected products available	Murphy et al., 2018
MARIUS WEATHER@HOME	25km, daily	UK, Three time slices (1900-2006, 2020-2049, 2070-2099)	Yes	100	One emission scenario (RCP8.5), one climate model, SST and sea ice sampling.	Bias correction required, bias corrected products available	Guillod et al., 2017

ATKINS STOCHASTICS DATASET	>200 locations, monthly	England and Wales, 1950-1997	Within regions	400	N/A	Bias correction applied to data	Atkins, 2020
AME STOCHASTICS DATASET	5km, daily	Greater Anglia, 1914-2017	Yes	1000	N/A	N/A	Dawkins et al., 2022

Table A3 | Description, key model characteristics and references of hydrological (HYD) and water resource system (WRS) models

	Model Name	Year	Spatial Repres.	Driving Input Time Series	Parameters In Model	Model Output (Time series)	Operations In Model	Access To Code	Refs
HYD	Grid-to-Grid	2009	Gridded	Precipitation, Potential evapotranspiration	5	Soil moisture, river flow	Abstractions (Rameshwaran et al. (2022)	No	Bell et al. 2009
HYD	DECIPHeR	2019	Semi-distributed	Precipitation, Potential evapotranspiration	7	River flow	-	Github	Coxon et al. 2019
HYD	GR4J/ GR6J	2017	Lumped	Precipitation, Potential evapotranspiration	4-6	River flow	-	R CRAN	Coron et al. 2017
HYD	CATCHMO D	1994	Semi-distributed	Precipitation, Potential evapotranspiration	5	River flow	Abstractions and effluent return	Python	Wilby et al 1994
HYD	SHETRAN- GB	2018	Gridded	Precipitation, Potential evapotranspiration	35	River flow	-	Email	Lewis et al. 2018b
W	Aquator	2008	Network of source/demand nodes	River flow, Water demand, environmental flows	182	Reservoir levels, energy/operatio n costs, abstractions	Water transfers, reservoirs, abstractions	Commercial software package	Oxford Scientific Software 2008
W	Pywr	2020	Network of source/demand nodes	River flow, water demand, environmental flows	11	Reservoir levels, energy/operatio n costs, abstractions	Water transfers, reservoirs, abstractions	Github	Tomlinson et al. 2020

	WREW	2020	Network of source/demand nodes	River flow, water demand, environmental flows	60	Reservoir levels, abstractions	Water transfers, reservoirs, abstractions	No	Dobson et al. 2020

Table A4 | Description, key characteristics, and references of hydrological projections

Data Product	Spatial And Temporal Resolution	Spatial And Temporal Coverage	Climate Projections	Hydrological And Groundwater Models	Outputs	Reference
FUTURE FLOWS	281 catchments, 24 boreholes Daily	Great Britain 1951-2098	UKCP09 Regional Climate Model Projections	Hydrological (CERF, PDM, CLASSIC), Groundwater (R-Groundwater)	River flows, groundwater levels	Prudhomme et al., 2013
EFLAG	200 catchments, 54 groundwater level boreholes and 558 groundwater bodies Daily	Great Britain, 1981-2080	UKCP18 Regional Climate Model Projections	Hydrological (Grid-to-Grid, GR4J, GR6J, PDM) Groundwater (Aquimod, ZOODRM)	River flows, groundwater levels and recharge	Hannaford et al., 2022
MARIUS DECIPHER	338 catchments Daily	England and Wales, 1975-2004, 2020-2049, 2070-2099	MaRIUS Weather@Home Climate Projections	DECIPHeR	River flows	Dobson et al., 2020
MARIUS GRID-TO-GRID	1km gridded Monthly	Great Britain, 2000-2006, 2020-2049, 2070-2099	MaRIUS Weather@Home Climate Projections	Grid-to-Grid	River flows and soil moisture	Bell et al., 2018

N: Novel indicators to inform drought management and water planning

Wilby[1], R.L., Murphy[2], C., O'Connor[2], P., Thompson[1], J.J. and Matthews[3], T.

[1]Geography and Environment, Loughborough University, LE11 3TU, UK

[2]Irish Climate Analysis and Research UnitS (ICARUS), Maynooth University, Ireland

[3]School of Geography, King's College London, WC2B 4BG, UK

Abstract

Indicators are important tools for tracking the socio-economic and environmental impacts of droughts as well as for checking progress in building resilience to future droughts under climate change. We begin with a synopsis of drought indicators used by the UK water sector to reduce disruption to supplies and harm to the environment. We explain how considerations of cost, immediacy, access, consistency, relevance, reliability, and others, have to be addressed when evaluating the suitability of information for new indicators. We then demonstrate the largely untapped potential of Google Trends (GT) search data and newspaper archives as resources for developing novel drought indicators in England and Ireland. We show that search terms such as "drought", "water butt" and "hosepipe ban" were significantly correlated with conventional hydroclimatic data during the period 2011-2022. There is also evidence of rising interest in water saving technologies and techniques, especially for outdoor water use. Meanwhile, search interest in Defra and the Environment Agency has declined and is more often associated with flood episodes than droughts. Similarly, interest in water companies in England is more likely to be associated with hosepipe bans than water leakage (although this varies by company). We discuss the implications of these findings for the development of public information campaigns, as well as for applying novel indicators to the monitoring of drought impacts in real-time.

1. Introduction

To measure is to manage – a principle that has been widely adopted by statutory- and non-statutory bodies alike. For instance, schools are graded on the basis of the quality of education provided, the behaviour, attitudes and personal development of pupils, amongst other criteria. Hospital quality indicators include the numbers of beds and types of service available, and nurse-to-patient staffing ratios. Railway companies are judged according to their safety and punctuality.

The UK 25 Year Environment Plan is framed in a similar way. The Defra (2020) Outcome Indicator Framework has 66 metrics covering 10 main themes. These span air and water quality, natural resources, resilience, natural beauty, biosecurity, resource use, and international dimensions. The resilience theme has three indicators, including for example "F3 Disruption or unwanted impacts caused by drought. This is about reducing the risks of harm from natural hazards and falls under the headline of resilience to natural hazards[1]." Table 1 reproduces details of this interim indicator and refers to the Supply Demand Balance Index (SDBI). This assesses how the supply-demand balance (water available for supply relative to forecast dry year demands) compares with what is set out in a water company's Water Resources Management Plan (WRMP) (Environment Agency, 2022). Water companies also have to provide related information about per capita consumption, outage, and leakage rates each year.

Table 1 Defra (2020) indicator F3: Disruption or unwanted impacts caused by drought. Key words and phrases are highlighted for the purpose of discussion.

Short description	**This indicator will focus on disruption to public water supply due to drought**. Water companies have a statutory duty to produce a water resources management plan (WRMP) and drought plan. The WRMPs, prepared, published and maintained in accordance with provisions of the Water Industry Act 1991 and regulations and directions made under it, must set out how a company intends to maintain the balance between supply and demand for water over at least the next 25 years. This includes how it will manage the increasing pressures on our water supplies from a growing population and climate change, whilst protecting the environment. Water company drought plans, also prepared, published and maintained under Water Industry Act 1991, set out the operational actions the water companies will take before, during and after a drought to maintain a secure supply of water. **This indicator will track changes in a Supply Demand Balance Index (SDBI)**, which will be reported by all water and sewerage companies from summer 2022. The SDBI will be reported within annual reviews of the WRMPs and as part of the Environment Agency's Environmental Performance Assessment (EPA) report.
Relevant goal(s) in the 25 Year Environment Plan	Reducing the risks of harm from environmental hazards
Relevant target(s) in the 25 Year Environment	Ensuring interruptions to water supplies are minimised during prolonged dry weather and drought. Boosting the long-term resilience of our homes, businesses and

[1] Whether there are any 'natural' hazards is now questionable given the extent of human-induced global change – but we shall overlook this oxymoron.

Plan	infrastructure
Position in the natural capital framework	Service/benefit
Related reporting commitments	Relevant to Sustainable Development Goals 11 and 13. Water and sewerage companies currently provide Security Of Supply Index (SOSI) data to the Environment Agency annually. This is published as part of the Environment Agency's EPA report and is part of the water companies' annual review of WRMPs.
Geographical scope	By **water company area** for those with customers wholly or mainly in England
Readiness and links to data	This indicator is not ready for reporting in 2020. SOSI data identifies whether water companies have a greater than planned risk of interruptions to public water supply during drought events. It illustrates those that need to take immediate action to increase resilience to the environmental hazard of drought. SOSI data is reported annually in the Environment Agency's annual EPA report. **From 2022, all water and sewerage companies will report a new, improved index (SDBI) annually. We will therefore use the SDBI as the metric for this indicator and report on it from 2023.**
Interim indicator (where applicable)	Not applicable

Other outcome indicators in the Defra (2020) framework cover aspects of freshwater and environmental quality but none of these explicitly refer to 'drought'. Nonetheless, drought was mentioned by 13 out of the 61 risks and opportunities in the third UK Climate Change Risk Assessment (CCRA3) (Table 2). The majority of these have a UK urgency score of 'More action needed'. This means that new, stronger or different Government action, whether policies, implementation activities or enabling environment for adaptation, over and above those already planned, are beneficial in 65 the next five years to reduce climate risks or take advantage of opportunities (Climate Change Committee, 2021a:9).

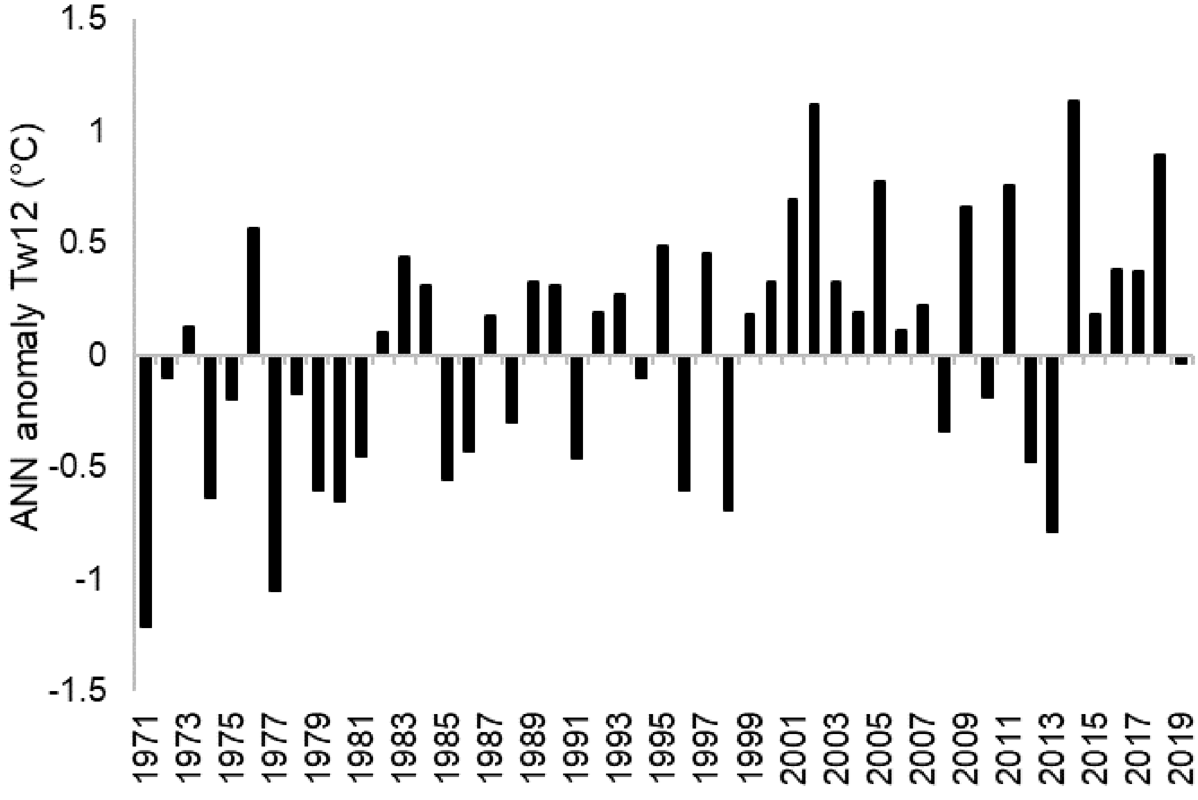
Table 2 Drought-related climate risks and opportunities identified by the CCRA3. Urgency is coded as: More action needed (red) or Further investigation (orange). Adapted from: Climate Change Committee (2021c).

Indicator	Link to drought
N6 Agricultural and forestry productivity	Decreased yields

N11 Freshwater species and habitats	Loss of habitats, reduced species abundance
N18 Landscape character	Changes in woodland, downland communities
I6 Hydroelectric generation	Reduced hydropower output and revenues
I8 Public water supplies	Reduced water availability
H1 Health and wellbeing	Wildfire and air pollution
H5 Building fabric	Shrink-swell of soils and subsidence
H10 Health	Public water supply interruptions
H11 Cultural heritage	Damaged buildings, archaeological sites, parks
H13 Delivery of education and prison services	Overheating of buildings
B3 Business production processes	Interruptions to water supplies
B6 Disruption to business supply chains and distribution networks	Insurance pay outs linked to drought indices
ID1 UK food availability, safety and quality	Decreased yields

The Climate Change Committee (2021b) used 49 indicators to Report Progress to Government of which eight were drought-sensitive or drought-related. These were: (1) condition of freshwater Sites of Special Scientific Interest in England; (2) status classifications of surface water bodies in England under the Water Framework Directive; (3) water temperature anomalies in Southern and Northern England (Figure 1); (4) weighted average water consumption per capita for households in England 2005-2020 and forecast to 2044-45; (5) proportion of properties with water meters from 1999-00 to 2019-20; (6) mid-century supply-demand balance for UK Water Resource Regions; (7) late-century supply-demand balance for UK Water Resource Regions; (8) total leakage for all water companies from 2000-01 to 2019- 20 against future commitments. Supporting analyses by ADAS (2019; 2021) confirm that the underpinning evidence base requires strengthening for many of these risks.

Figure 1 Water temperature (Tw) index for Northern (left) and Southern (right) England showing annual variance from the long-term mean. Sources: Climate Change Committee (2021); Wilby and Johnson (2020)



CCRA3 refers to the National Water Framework for Water Resources because it sets out seven steps to better understand and prepare for water needs to 2050 (Box 1). As highlighted, these will require information on the incidence of rota cuts and standpipes; amounts of water abstraction; per capita water usage; leakage rates; use of drought permits and orders; increased water supplies; and water transfers. Others have developed indicators of physical climate risks to the UK, including the proportion of time under severe hydrological drought (Arnell et al., 2021) based on standard drought indices (Bachmair et al., 2016a; Hannaford et al. 2023, this issue). However, relatively little attention has been given to the socio-ecological responses to present and projected droughts (Wilby, 2020). An indicator on 'Total annual spend on resilience measures by all water companies' was reviewed by ADAS (2019) but not included in the final set of metrics reported by the Climate Change Committee (2021b). Moreover, it has been noted that "…while impacts are often used to define drought, in the UK this is usually undertaken in hindsight rather than actively monitored and reported publicly…" (Hannaford et al., 2019:56).

Box 1 Role of regional planning in the National Water Framework for Water Resources. Source: Environment Agency (2020:6)

Increasing resilience to drought – so that restrictions such as rota cuts and standpipes are needed no more than once every 500 years on average by the 2030s

Delivering greater environmental improvement – by considering changes to water abstractions, beyond those already identified in water company plans, that will deliver a sustainable abstraction regime across all sectors

Long-term reductions in water usage – through adopting a planning assumption of achieving, on average, 110 litres per person, per day, of water use by 2050 while also reducing non-household demand

Leakage reduction – through delivery of the industry's target to reduce leakage by 50% by 2050

Reducing the use of drought permits and orders – by understanding the environmental risk of each measure and using them less frequently, particularly at sensitive water sources or habitats

Increasing supplies – by exploring a range of options, such as reservoirs, water reuse schemes and desalination plants; considering the potential for developing and sharing supplies with other sectors; and through work to improve water management in catchments

Moving water to where it's needed – by fully exploring all opportunities for water transfers, within and between regions, of different scales and lengths.

Hence, there are gaps in national capability to (1) track impacts of past and present droughts; as well as (2) monitor progress in building resilience to future drought risks. This paper considers how novel indicators – based on print media and online searches – could be deployed to meet these challenges and support water planning. The next section begins by setting out the desirable properties of a (drought) risk indicator. This is followed by an appraisal of recent advances in the use of data from online searches to monitor spatiotemporal awareness of drought. We then review the emerging use of newspaper and documentary analysis to uncover past droughts and their impacts. These two elements are bought together in a pilot study of coincident search terms and drought impacts in England and Ireland within the period 2011 to 2022. These countries were chosen for comparative purposes and to limit the possibility of contrasting drought/non-drought conditions over larger areas (such as the UK as a whole). Finally, we discuss future possibilities for developing more nuanced indicators of drought impacts and societal awareness, in near real-time.

2. Novel indicators of drought

According to ADAS (2021) indicators can be used to assess trends in (1) risk factors (hazard, vulnerability and exposure); (2) impacts (across various sectors and administrative units); (3) resilience and adaptation actions (input and output). As noted before, some drought risk factors and a few climate change impacts are covered by existing indicator sets (Table 2). However, less is known about the attendant social and environmental impacts and responses to droughts.

When developing new hydroclimatic indicators, it is helpful to refer to a checklist of desirable properties (Table 3). Ideally, the underpinning data are benchmarked in some way to enable consistent comparisons over time and space. Climate risk or resilience indicators should be sensitive to primary hydro-meteorological variables such as temperature, rainfall, solar radiation, atmospheric humidity and wind speed. The link should be direct and physically-sensible, without intermediate or confounding factors. Data should be freely available, routinely and quickly updated, plus economically feasible to sustain over decades. Long-term, quality assured indicator series are needed to discern emergent trends within (often) 'noisy' data sets. Finally, indicators should be interpretable by decision-makers and planners, as well as resonate with public interest. By satisfying these conditions, there is scope for the publication of indicators that can track changes in drought impacts and awareness over time, as well as offer useful insights for their management.

Table 3 Desirable properties of climate risk and resilience indicators. Informed by: 603 DETR (1999) and Ekström et al. (2018).

Property	Explanation
Benchmarked	Changes in the indicator are always compared with the same reference case, such as a baseline reporting period of 1961-1990.
Sensitive to climate	Variations in the indicator are associated with physically-plausible climate drivers, such as between air and river water temperatures.
Relevant to decisions	The indicator describes conditions that are relevant to planners, such as changes in per capita water consumption over time.

Based on long series	Long series are helpful for distinguishing between short-term variations and long-term trends, as with the NHMP Outflow Series.
Based on open data	Data used to develop and update indicators are openly and freely available, such as those from the CEH UK Water Resources Portal.
Based on reliable data	Data used to support the indicator are quality assured with good meta data, such as groundwater levels from the British Geological Survey.
Readily updated	Data underpinning the indicator are routinely updated with limited latency, such as HadUKP which is published within just a few days.
Affordable to maintain	Costs are low/ economically justified for long-term data collection and reporting, such as for routine water resource monitoring and outlooks.
Few confounding factors	Variations in the indicator are driven primarily by climatic factors, such as heightened demand for irrigation during periods of low rainfall.
Public resonance	Variations and trends in the indicator raise awareness or chime with public concerns, such as around water distribution leakage rates.

2.1. Analysis of online search terms

Google Trends (GT) is a publicly available tool that allows users to analyse the relative popularity of search terms over time. It provides data on the number of Google searches for a particular term that can be used to track changes in search volume for any topic. The data are anonymized, categorized and aggregated allowing users to gauge interest in search terms or topics at global to city-scales (assuming sufficient search volumes are available). Providing absolute search information would return billions of entries every day which would not be feasible to describe. Therefore, data are presented as a proportion of all searches on all topics on Google for the specified period and geographic unit, in a way that accounts for the changing numbers of interest users through time. GT has two filters for real-time and historical datasets. Real-time gives searches covering the past seven days, compared with non-real time, which is a sample of the entire Google dataset from year 2004 to 36 hours ago. This offers scope for following public interest in droughts and their impacts in near real-time (as in de Brito et al., 2020).

However, GT does have limitations. Although data aggregated by GT are referenced by geographical area, searches made by users may be for events that are taking place or have occurred in places remote from the physical location of the internet browser. For example, by using Google to search for information about “drought” in Australia, a Loughborough-based user would add to England counts of searches on the topic of drought. In other words, counts reflect the geographic location of the searcher rather than the search topic. Hence, this effect may confound associations. Moreover, spikes in search volumes for drought could be driven by other factors such as media coverage in another country or totally unrelated issues, for example, England-based searches for the lyrics of the Beyoncé song ‘Love Drought’.

Despite these recognized limitations, there has been much uptake of GT in hydroclimatic research. For instance, Kam et al. (2019) used GT to analyse decay patterns in drought interest, whereas Kim et al. (2019) tracked drought awareness at regional and national scales across the United States by comparing search interest with data from the US Drought Monitor. Similarly, Park et al. (2022) found that spikes in public interests in drought depend on drought intensity. Others have used GT to investigate growing public health concerns about micropollutants in waste, surface or ground water in the UK (Mavragani et al., 2016); or around heat-health vulnerabilities in Ireland (Paterson and Godsmark, 2020). At the other hydrological extreme, Thompson et al. (2020) demonstrated that GT could help evaluate flooding in places (Kenya and Uganda) where formal hydro-meteorological data are scarce.

2.2. Analysis of newspaper articles

There is growing interest in the use of newspapers archives (Dayrell et al., 2022; Murphy et al., 2017; Noone et al., 2017; van der Schrier et al., 2021), documentary sources (Brázdil et al., 2018), and social media (Antwi et al., 2022) to investigate past droughts and their impacts. Such techniques have even enabled the (re)discovery of otherwise forgotten extreme events (e.g., Murphy et al., 2020).

The Irish Drought Impacts Database (IDID) (Jobbová et al., 2023) provides a catalogue of historical drought impacts compiled from print media covering the period 1733-2019. The IDID comprises more than 11,000 drought impact reports, identified and categorised through systematic searches of the Irish Newspaper Archives (INA) [2] using key terms such as “drought” and “droughts”. The INA provides a good sample of national and regional print media from at least 50 titles annually covering the majority of Irish counties but is not an exhaustive database of all Irish newspapers. Identified newspaper reports were individually examined to remove irrelevant articles, such as those referring to the surname “Drought” or a “scoring drought” in sporting terms. Remaining articles were then grouped into 15 drought impact categories using a modified version of the impact categories developed as part of the European Drought Impact Inventory (Stahl et al., 2012). This includes agriculture and livestock farming, forestry, energy and industry, tourism and recreation, public water supply (PWS), water quality, freshwater ecosystem (habitats, plants and wildlife), terrestrial ecosystem (habitats, plants and wildlife), soil systems, wildfires, air quality, and human health. The date of impact, date of report, newspaper title, associated drought categories and sub-categories, location of impact (local area, county, region), timing of impacts and relevant quotations from the article are provided for each impact (where available) in the IDID. Hence, the dataset provides detailed information

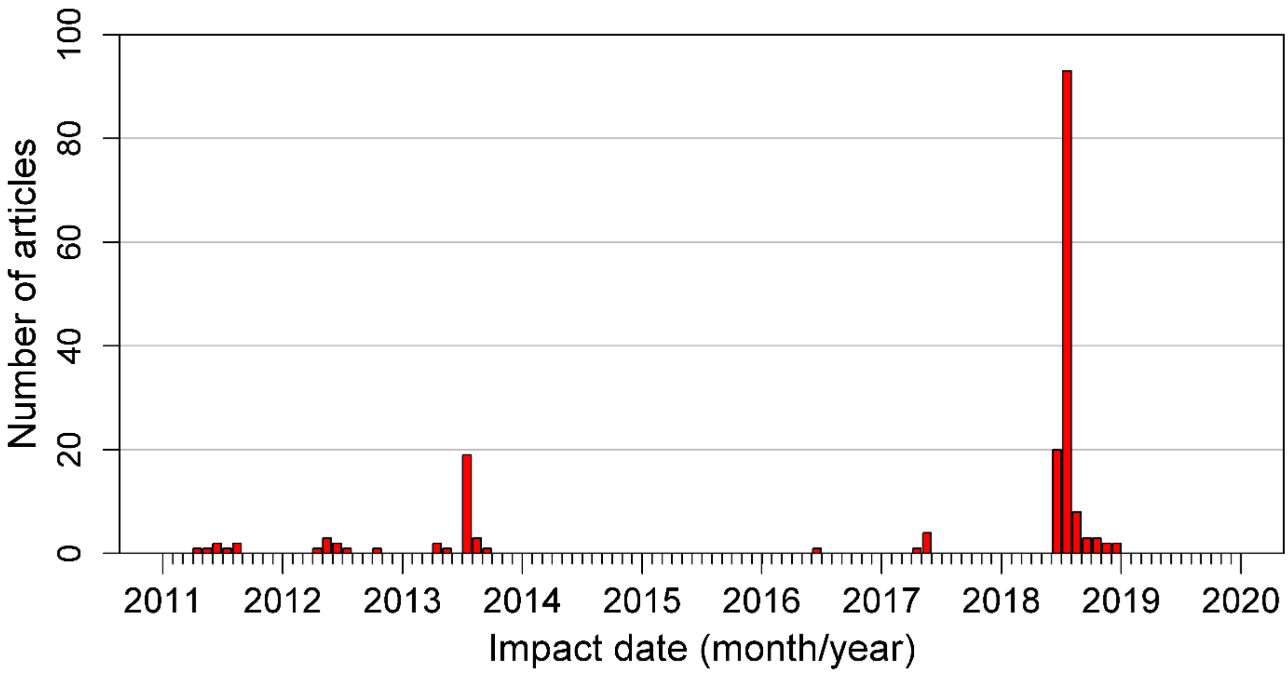
[2] https://www.irishnewsarchive.com/

about the temporal and geographic extent of drought events, their socio-economic context, impacts and responses.

Although the IDID provides valuable information on drought impacts the database has several limitations (Jobbová et al., 2023), including changing frequencies of publication, variations in life span and spatial coverage of newspaper titles in the INA. For our period of analysis, the annual number of newspapers in the INA remains stable at more than 60 titles. However, a predominance of regional newspapers in some counties and larger centres of population may create biases through increased reporting in these areas. While the IDID includes Northern Ireland, there are fewer titles there relative to the Republic. More broadly, use of newspaper records for investigating drought may be prone to other biases such as reporting of impacts subject to editorial judgements about what is newsworthy at the time and alongside other stories. The number of impact reports also varies markedly between impact categories and is dominated by articles about agriculture and livestock and public water supplies. No doubt this reflects the importance of these sectors in Ireland, but the focus also reflects what is considered as important from a journalistic perspective so should not be considered a comprehensive representation of wider drought impacts. Despite these limitations the IDID has been shown to closely relate to quantitative meteorological and hydrological drought metrics in Ireland (O'Connor et al., 2022a;b; Jobbová et al., 2023).

Here, we use impact reports from the IDID for the period concurrent with GT data (2011-2019). This includes 178 reports across the impact categories (Figure 2). The significant impact of the summer 2018 drought is immediately apparent from the spike in reports during this period. Overall, the most frequent impacts were recorded for agriculture and livestock and PWS categories, with the most frequently recorded sub categories, representative of the type of impact identifiable in the given article, being reduced crop productivity for the former and local water supply shortages for the latter. Notably, nine categories had fewer than 10 impact reports so were excluded from further analysis.

Figure 2 Total monthly IDID drought impact reports, by month of impact, across all categories during the period 2011-2019.



3. Pilot study

3.1. Data and methods

Here, we demonstrate the application of GT and a newspaper archive to the analysis of societal and environmental impacts of droughts in England and Ireland. Following Hannaford et al. (2019), we are particularly interested in the feasibility of using these resources as near real-time (rather than retrospective) indicators of drought impacts. Others have also called for “a better framing of drought as a coupled dynamic between the environment and society”, with greater emphasis on indicators for drought monitoring and in early warning systems (Bachmair et al., 2016b: 516).

GT allows the comparative analysis of up to five search terms. We began by exploring detailed variations in search interest for broad terms such as “drought”, “heatwave”, and “hosepipe” over the period 2018 to 2022 as these years bracket three notable events (in 2018, 2020, and 2022) across the GT geographic domains of England and Ireland. We then evaluated a set of 25 terms, clustered into five thematic groups, each with five terms (Table 4). For example, our ‘Household’ group has search terms: “dual flush toilet”, “grey water”, “tap aerator”, “water butt”, and “watering can”. Our initial choice of terms was informed by trial and error, as well as by reference to previous studies (e.g., Lee et al., 2022). Terms considered but subsequently excluded due to small sample sizes were: “bottled water”, “desalinisation”, “drought index”, “Drought Order”, “river flow”, “runoff”, “soil moisture”, “water abstraction”, “water cycle”, “water deficit”, “water demand”, “water pricing", and "water stress". Other terms – such as "river level" – were excluded because they more often associated with floods than droughts.

Table 4 Groups of search terms. Those with greatest interest in each group during 2018 to 2022 are highlighted in bold. Note that Group 2 organisations were changed for equivalent organisations in Ireland.

Group 1 Households	Group 2 Organisations	Group 3 Impacts	Group 4 Regulation	Group 5 Hydrological
Dual flush toilet Grey water Tap aerator **Water butt** Watering can	Defra **Environment Agency** Ofwat SEPA Water company	Algae bloom Heath fire Subsidence **Water leak** Water shortage	**Hosepipe ban** Water customer Water meter Water supply Water use	Groundwater Infiltration Irrigation system Water evaporation **Water management**

Monthly GT search interest data were then downloaded separately for the 25 terms in Table 4 for the period 2011 to 2022, for England and Ireland separately. This ensures that each term is self-calibrated within the range 0 (no search interest) through to 1 (the month with greatest search interest). Extracted series were detrended by the method of differencing, whereby a new series is created by taking the difference in GT values between successive months. These detrended series were then cross-correlated with others in the group, across the set as a whole, as well as with two hydroclimatic indices for England (Met Office HadUKP monthly precipitation series[3] and the National Hydrological Monitoring Programme [NHMP] Outflow Series[4]) as well as the newspaper report counts for Ireland. Critical values for the Pearson correlation coefficient (rcrit) were adjusted using the Bonferroni correction given the large number of correlation tests performed. For instance, when performing 50 separate correlation tests, rcrit increases from 0.16 (p=0.05) to 0.27 (p=0.001) for a sample size N=144 months and a two-tailed test.

The same 25 GT interest search terms were assessed for the domain of Ireland but the England government agencies listed in Group 2 were replaced with Irish equivalents (i.e.

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[3] https://www.metoffice.gov.uk/hadobs/hadukp/data/download.html

[4] CEH National River Flow Archive, pers. comm.

Department of Environment, Environmental Protection Agency [EPA], Commission for Regulation of Utilities [CRU], Northern Ireland Environment Agency [NIEA] and Irish Water) for the period 2011-2019. Counts of monthly drought impact reports – based on date of impact rather than of publication – were extracted from the IDID for the most common impact categories (i.e., agriculture and livestock, PWS, freshwater ecosystems, and wildfires). Both datasets were detrended using the method of differencing, and resultant series cross-correlated. The GT search terms that correlated significantly with the selected IDID categories were then identified, firstly using the rcrit value of 0.19 (p=0.05) and then a more stringent (p=0.001) Bonferroni corrected rcrit value of 0.33 (sample size N=108; two-tailed test). All four categories showed significant correlations with at least one GT interest term, with the most strongly correlated terms being "hosepipe ban", "water shortage" and "water butt" (in that order). Detrended series for these GT search terms were subsequently plotted against impact reports to visualize their association.

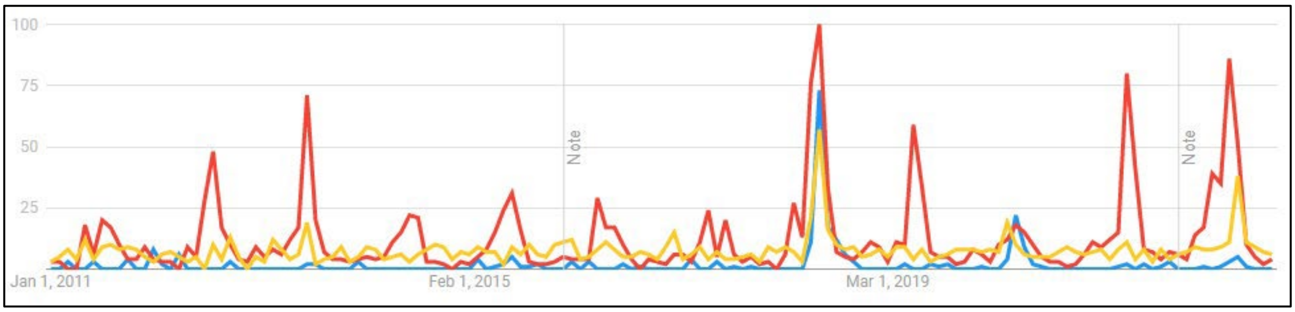
3.2. Results

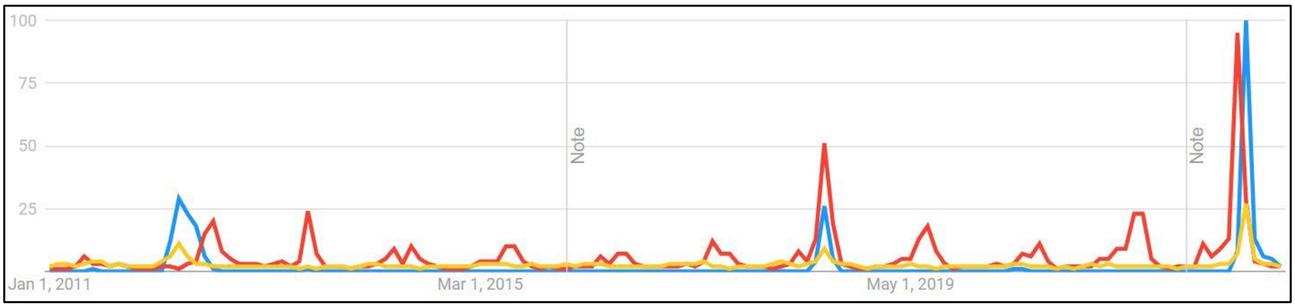
3.2.1. Overview

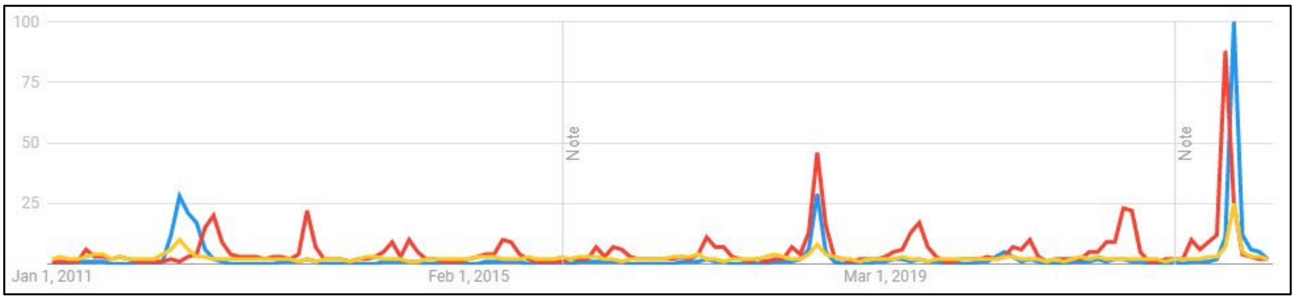
The findings of the GT and newspaper archive analysis are presented in three stages. First, the relative interest and intra-annual phasing of key search terms are described. Second, multi-annual trends and correlations amongst terms are presented. Third, the strongest associations between GT interest versus hydroclimatic indices for England and droughts impacts in Ireland were identified.

Over the periods 2018–2022 and 2011–2022 there was more interest in "heatwave" than "drought" with search volumes 69-72% and 28-31% of the totals respectively in the four nations of the UK (compared with a more even divide of 42-60% and 40-58% across regions of Ireland). Interest in "hosepipe ban" in England spiked in April 2012, July 2018, and August 2022, coinciding with actions taken by water companies to preserve stocks at those times (Figure 3). The largest spike in the Ireland series occurred in July 2018 – the month when a state of absolute drought was declared in the Republic because there had been no rainfall at the vast majority of weather stations in the previous 14 days.

Figure 3 GT interest in search terms "drought" (yellow line), "heatwave" (red line), and "hosepipe" (blue line) in Ireland, England (and the UK) during the years 2011 to 2022, having normalized across the period as a whole. Note that GT system changes were made to data collection in 2016 and 2022. (Top to bottom: Ireland, England, UK)







3.2.2. Droughts in England

There were subtle variations in the timing of peak interest within and between years (Figure 4). In England, this was most marked during summer 2022 when interest in the record-breaking heatwave[5] peaked in 17-23 July, a few weeks before greatest interest in drought (and hosepipe bans) in 7-13 August. Conversely, in 2018 and 2020, peak interest in drought (and hosepipe bans) preceded peak interest in the heatwave due to exceptionally dry conditions in June and spring respectively. Hence, the GT data appear capable of detecting and discriminating variations in the temporal evolution of the two hazards between years.

Other trends emerge over the longer-term amongst some of the 25 search terms (Figure 5). Overall, “hosepipe ban” (Group 4) attracts most interest but this is highly episodic. Within other groups the most popular terms were “water butt” (Group 1), “Environment Agency” (Group 2), “water leak” (Group 3), and “water management” (Group 5). Strong upward trends are evident in the volume of searches for all household terms (Group 1), but most notably for “dual flush toilet”, “grey water” and “tap aerator”. Interest in “water butt” and “watering can” increased too but is highly seasonal with clear summer maxima. Search interest in organisations (Group 2) is generally waning with notable declines for “Defra” and the “Environment Agency”. However, intervening spikes of interest during winters 2013/14, 2015/16 and 2019/20 coincided with major flood episodes. Interest in impacts (Group 3) has risen most for “water leak” but there are also spikes in search volumes for “water shortage”. All regulation terms (Group 4) show a strong increase

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[5] A new record temperature of 40.3 °C was set on 19 July 2022 at Coningsby, England.

(except “hosepipe ban”) with significantly more interest in “water use” and “water meter” in 2022 than in 2011. Search volumes for hydrological terms were comparatively low and noisy apart from “irrigation system” which is highly seasonal and typically peaks in May to July.

Within groups, there were significant ($p = 0.001$) correlations amongst detrended search data for “water butt” versus “watering can” ($r = +0.60$); “water leak” versus “water shortage” ($r = +0.52$); “water meter” and “water use” ($r = +0.52$); and “infiltration” versus “water evaporation” ($r = +0.47$). Other significantly correlated pairs of terms were: “grey water” versus “water butt” ($r = +0.55$); “heath fire” versus “water shortage” ($r = +0.46$); “hosepipe ban” versus “water use” ($r = +0.48$); and “groundwater” versus “evaporation” ($r = +0.39$). Across all search terms, the strongest correlations were for “hosepipe ban” versus “water butt” ($r = +0.72$); “hosepipe ban” versus “water company” ($r = +0.69$); “water shortage” versus “water supply” ($r = +0.65$); and “watering can” versus “irrigation system” ($r = +0.65$). However, inspection of these associations reveals they are dominated by a few outliers (Figure 6). For example, when there is high interest in hosepipe bans people are also searching for information about water butts – presumably as an adaptation measure. Similarly, the strong association between watering can and irrigation system could reflect concerns about dry soils and damage to garden plants/ crops.

Figure 4 As in Figure 3 but for England during years 2018 to 2022, normalized over this period as a whole. Note the data collection system changed from 1/1/22. (Top to bottom: 2018, 2019, 2020, 2021, 2022, 2018-2022).

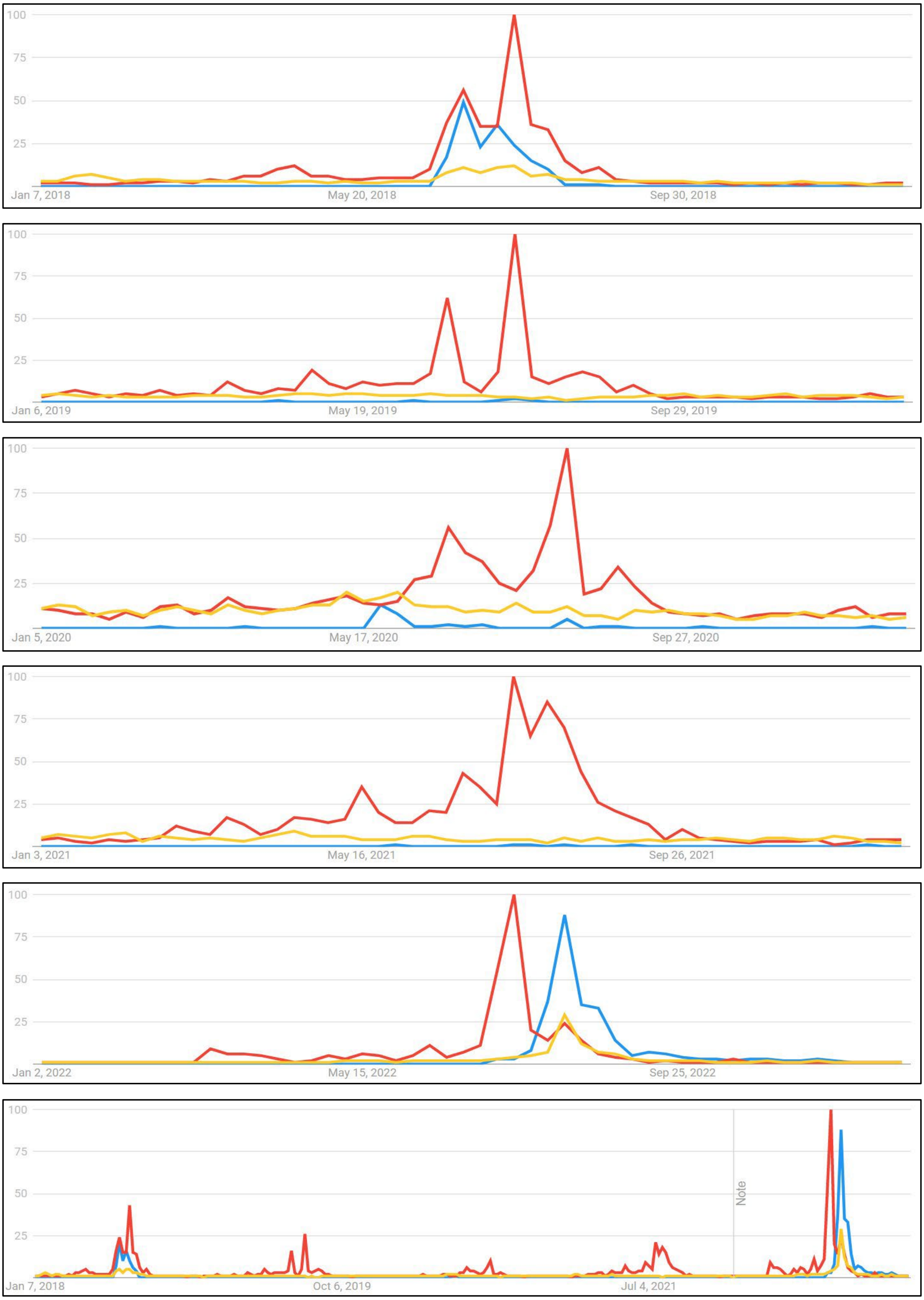
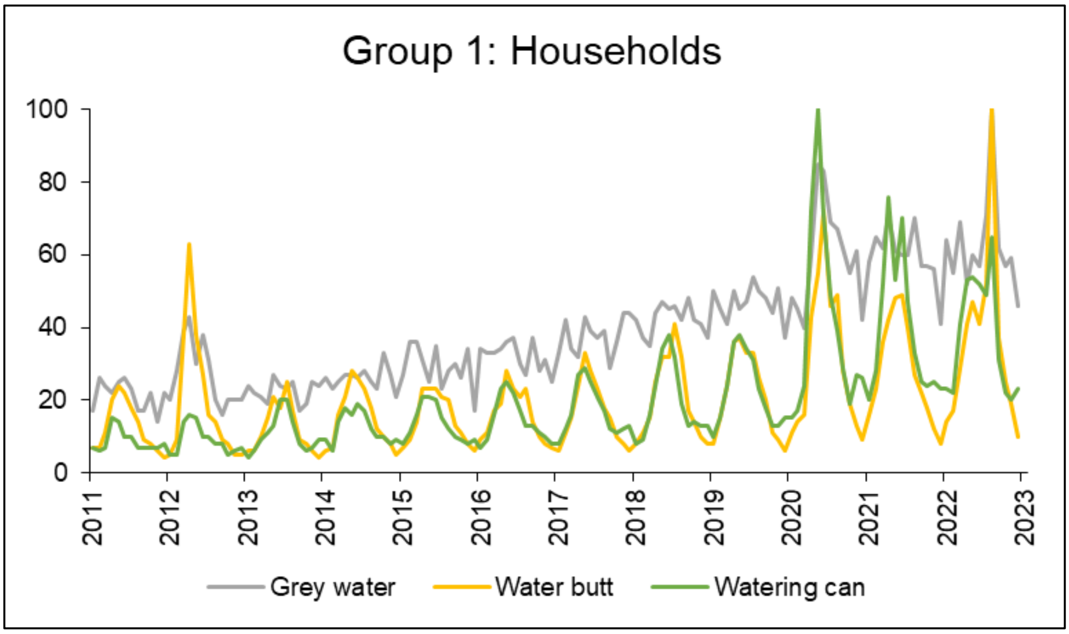
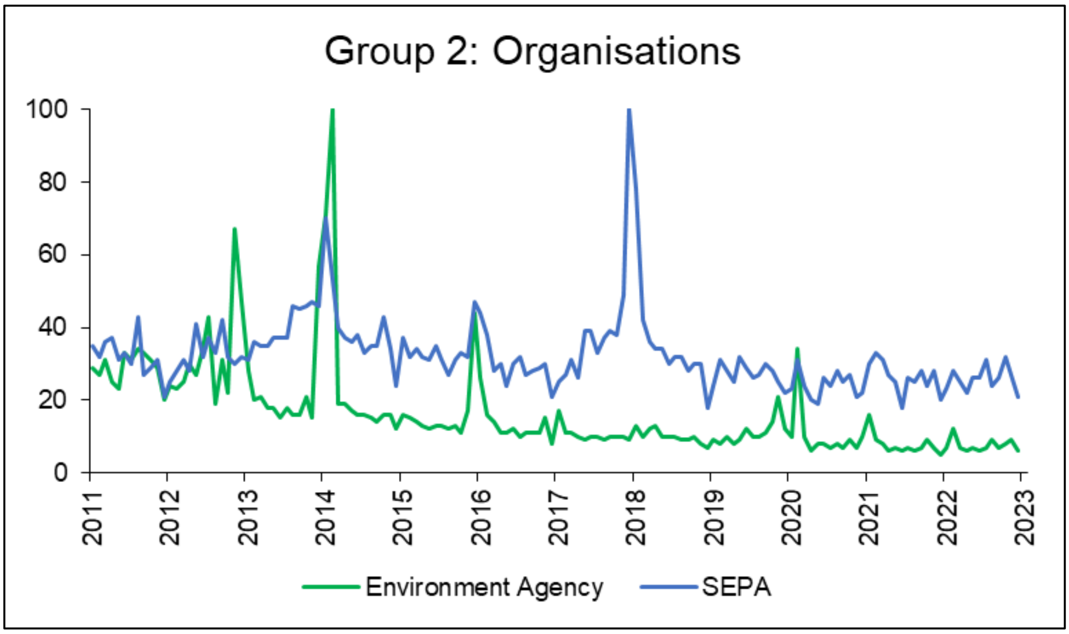
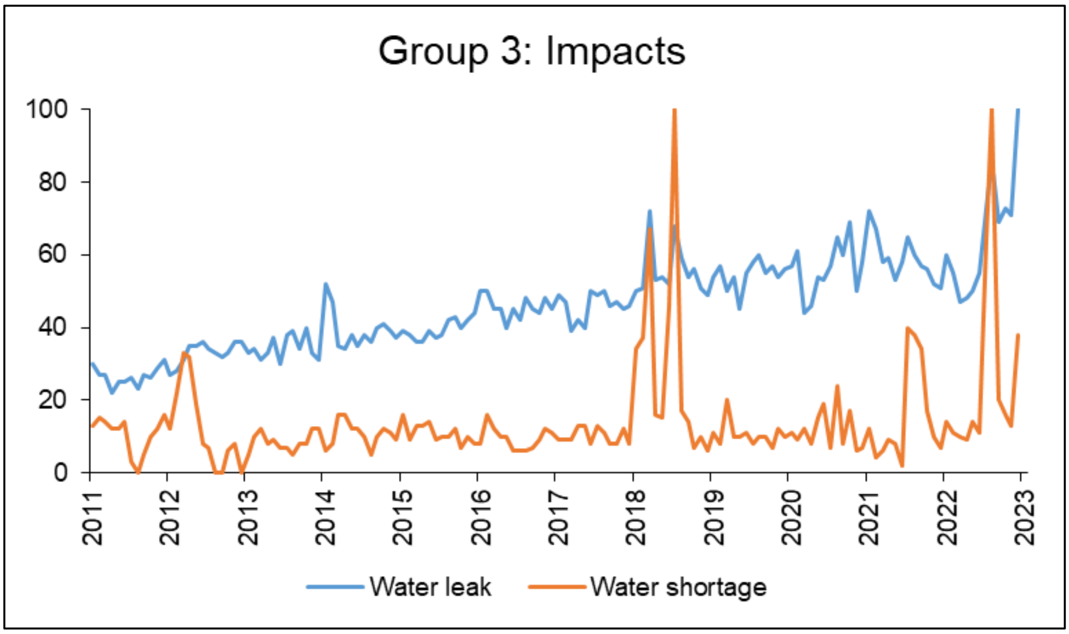


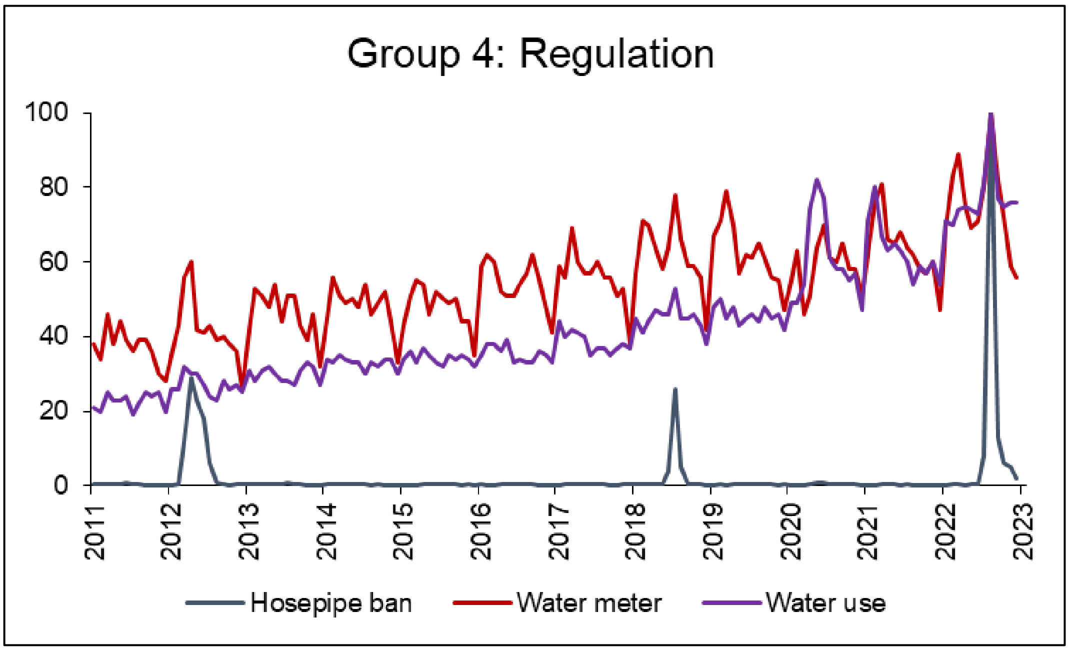
Figure 5 Illustrative trends and associations amongst GT search volumes around "households" (grey water, water butt, watering can), "organisations" (Environment Agency, SEPA), "impacts" (water leak, water shortage), "regulation" (hosepipe ban, water supply), and "hydrological" (irrigation system, water management) terms in England over the period 2011 to 2022. Changes in comparative volumes are also given for the most popular terms in each group (lower right panel).



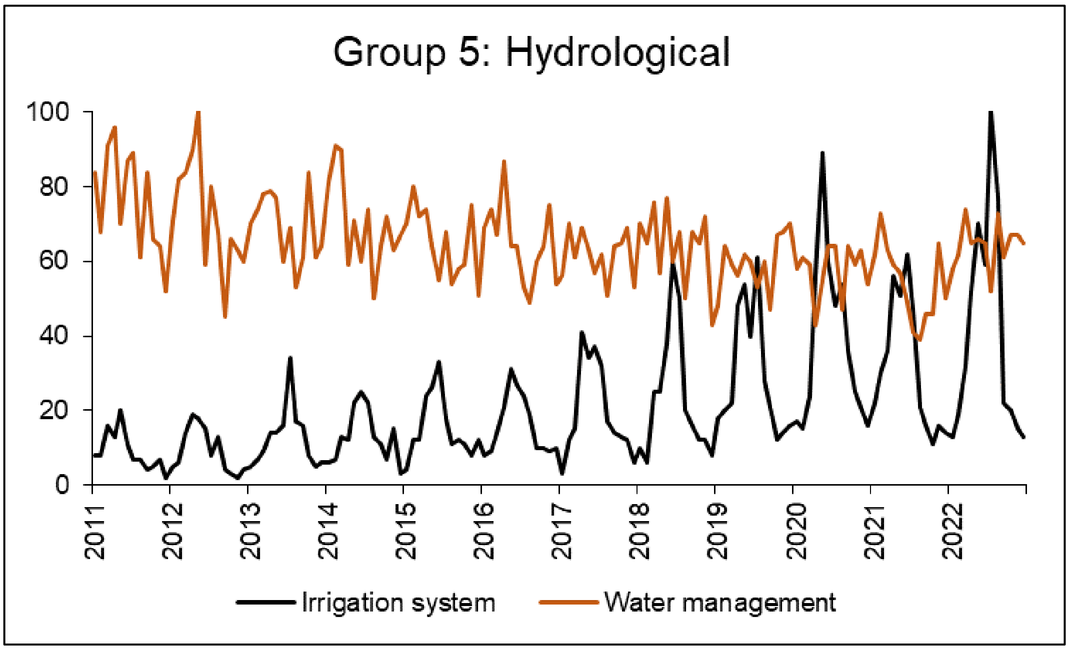




Group 3: Impacts
100
80
60
40
20
0
2011
2012
2013
2014
2015
2016
2017
2018
2019
2020
2021
2022
2023
Water leak
Water shortage



Group 4: Regulation
100
80
60
40
20
0
2011
2012
2013
2014
2015
2016
2017
2018
2019
2020
2021
2022
2023
Hosepipe ban
Water meter
Water use


Group 5: Hydrological
100
80
60
40
20
0
2011
2012
2013
2014
2015
2016
2017
2018
2019
2020
2021
2022
Irrigation system
Water management

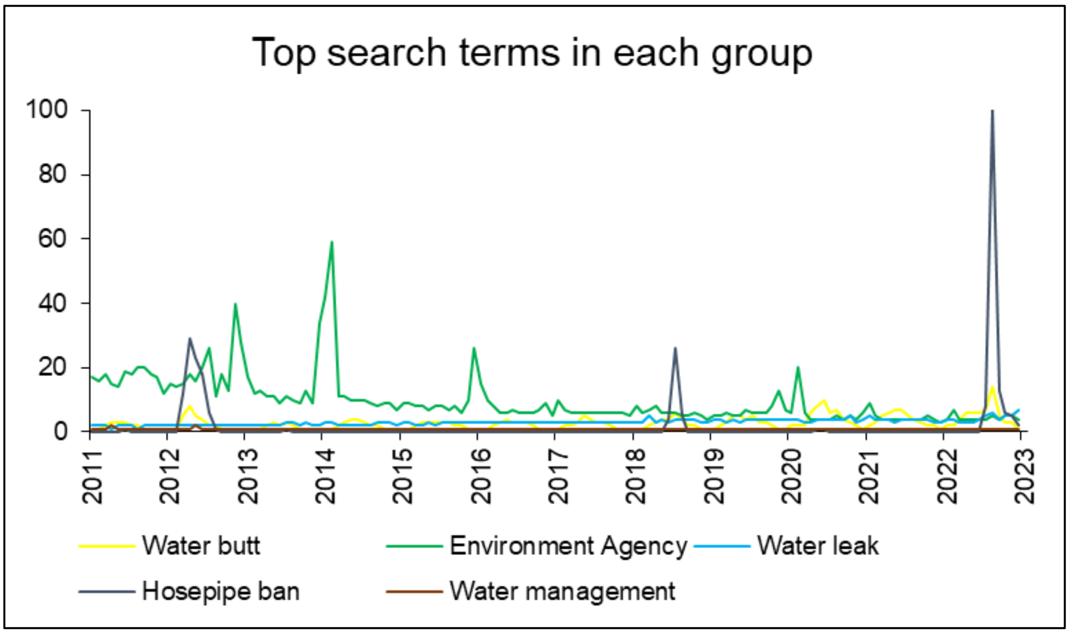
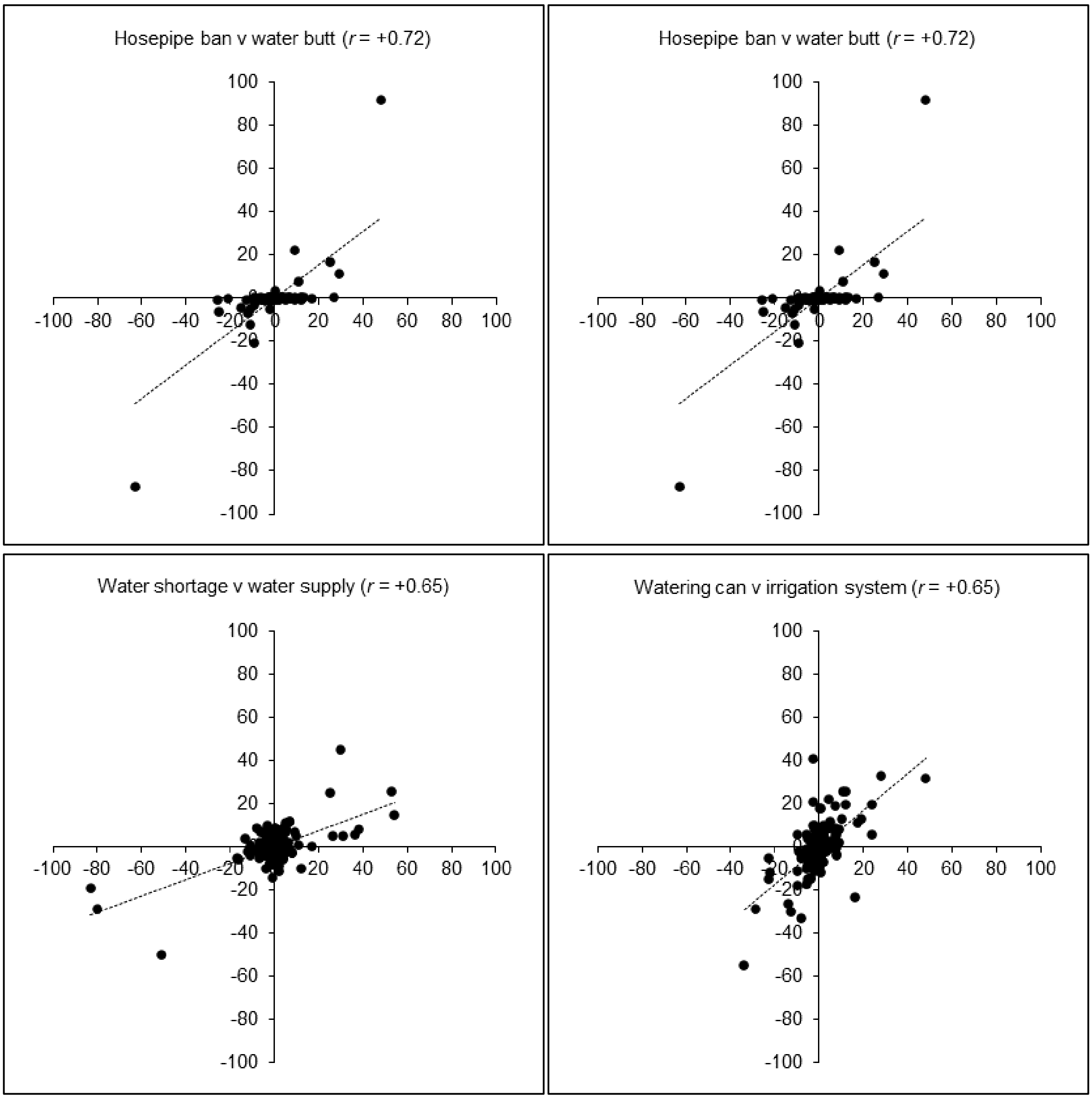

Top search terms in each group
100
80
60
40
20
0
2011
2012
2013
2014
2015
2016
2017
2018
2019
2020
2021
2022
2023
Water butt
Environment Agency
Water leak
Hosepipe ban
Water management

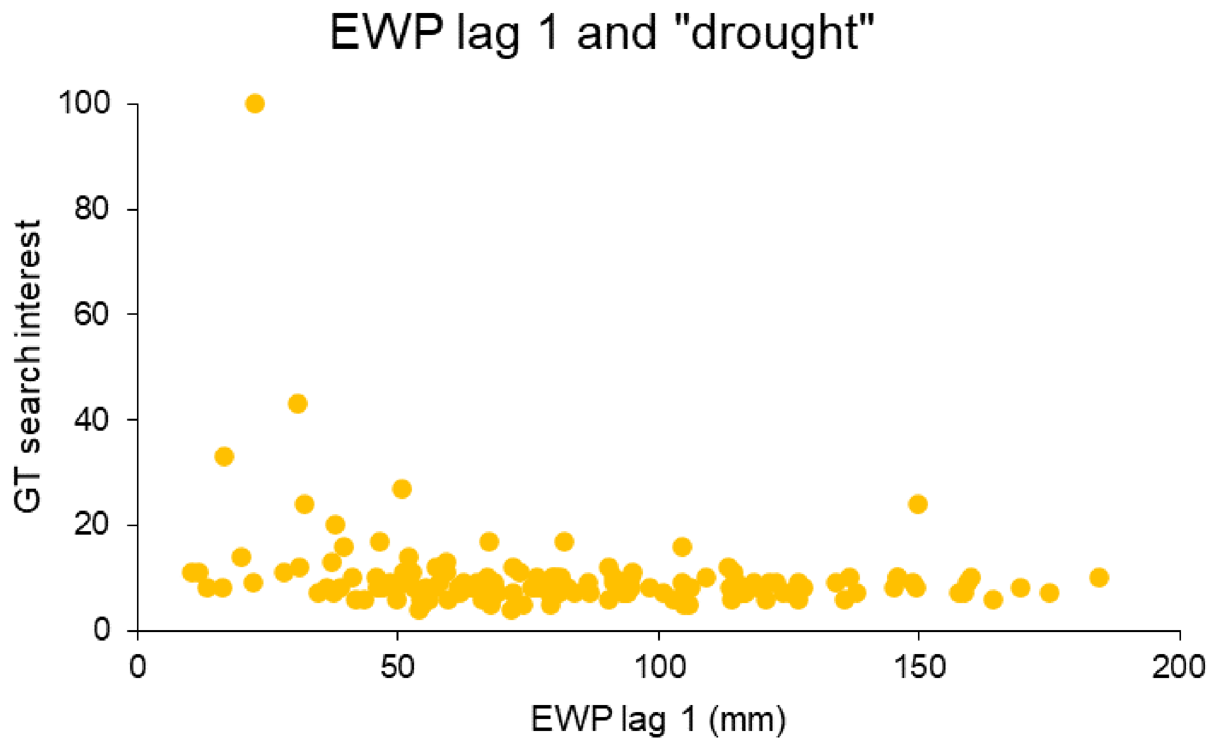
Figure 6 Correlations (r) in GT interest amongst drought-related search terms for the UK over the period 2011 to 2022. All data have been detrended.

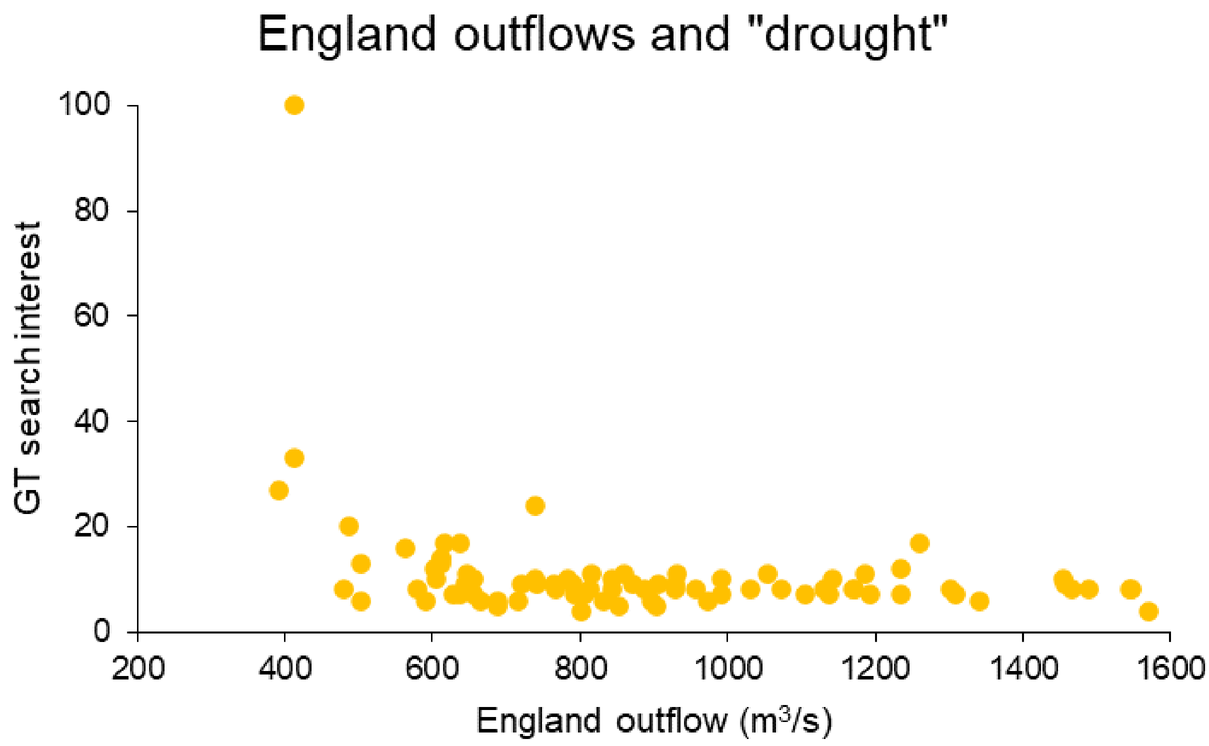


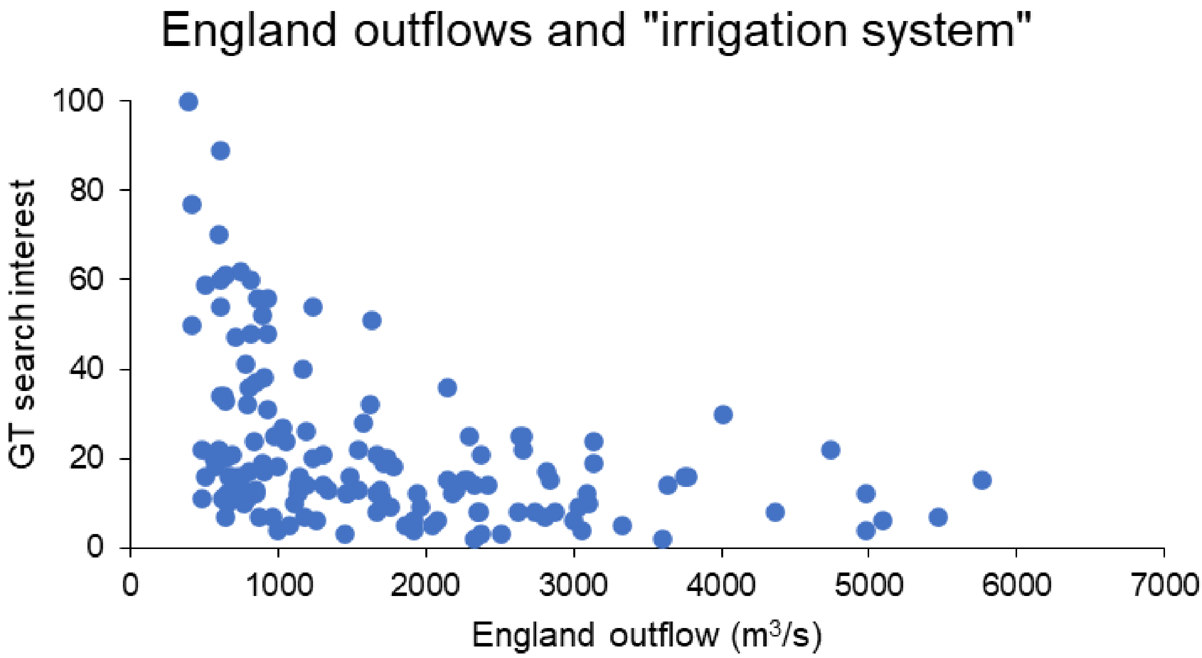
Next, we investigated associations between GT search volumes, hydrometric indices for England (coincident and lagged in time), and documented impacts in Ireland. The strongest negative correlations were between lag-1 monthly England and Wales Precipitation (EWP) and “water butt” (r = -0.42), “watering can” (r = -0.28), “heath fire” (r = -0.28), and “irrigation system” (r = -0.31); the only significant positive correlation was for “Environment Agency” (r = +0.27). The strongest negative correlations with the non-lagged monthly England Outflows series were also for “water butt” (r = -0.45), “watering can” (r = -0.32), “heath fire” (r = -0.23) and “irrigation system” (r = -0.40). The strongest positive correlations were for England outflows and “Environment Agency” (r = +0.51), and “groundwater” (r = +0.38).

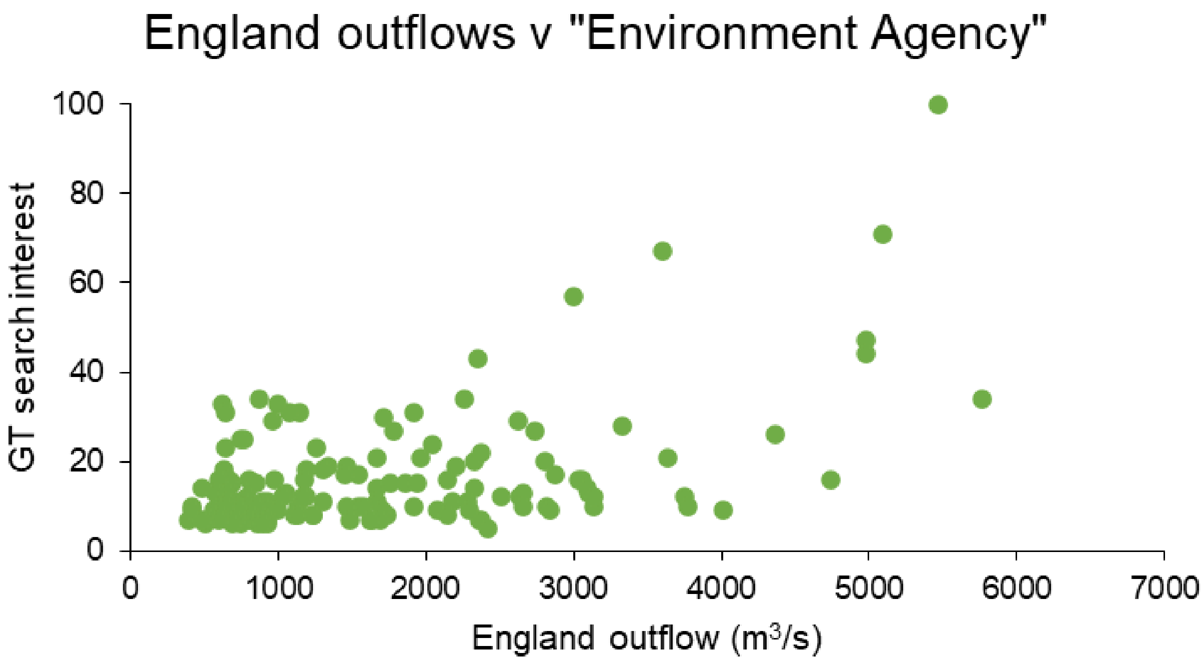
Inspection of the data for selected search terms suggests non-linear associations with the rainfall and runoff series (Figure 7). For instance, search volumes for “drought” increase markedly when the EWP total for the previous month is <50 mm, or when England outflows for the concurrent month are <500 m3/s. Interest in “irrigation system” falls dramatically when the monthly mean outflow is >1000 3/s. As noted previously, greatest interest in the “Environment Agency” is during episodes of high outflow.

Figure 7 Associations between EWP (lag 1) and England outflows versus GT search interest in “drought”, “irrigation system”, and the “Environment Agency” during the period 2011 to 2022.


EWP lag 1 and "drought"
GT search interest
EWP lag 1 (mm)
0
20
40
60
80
100
0
50
100
150
200


England outflows and "drought"
GT search interest
England outflow (m3/s)
0
20
40
60
80
100
200
400
600
800
1000
1200
1400
1600



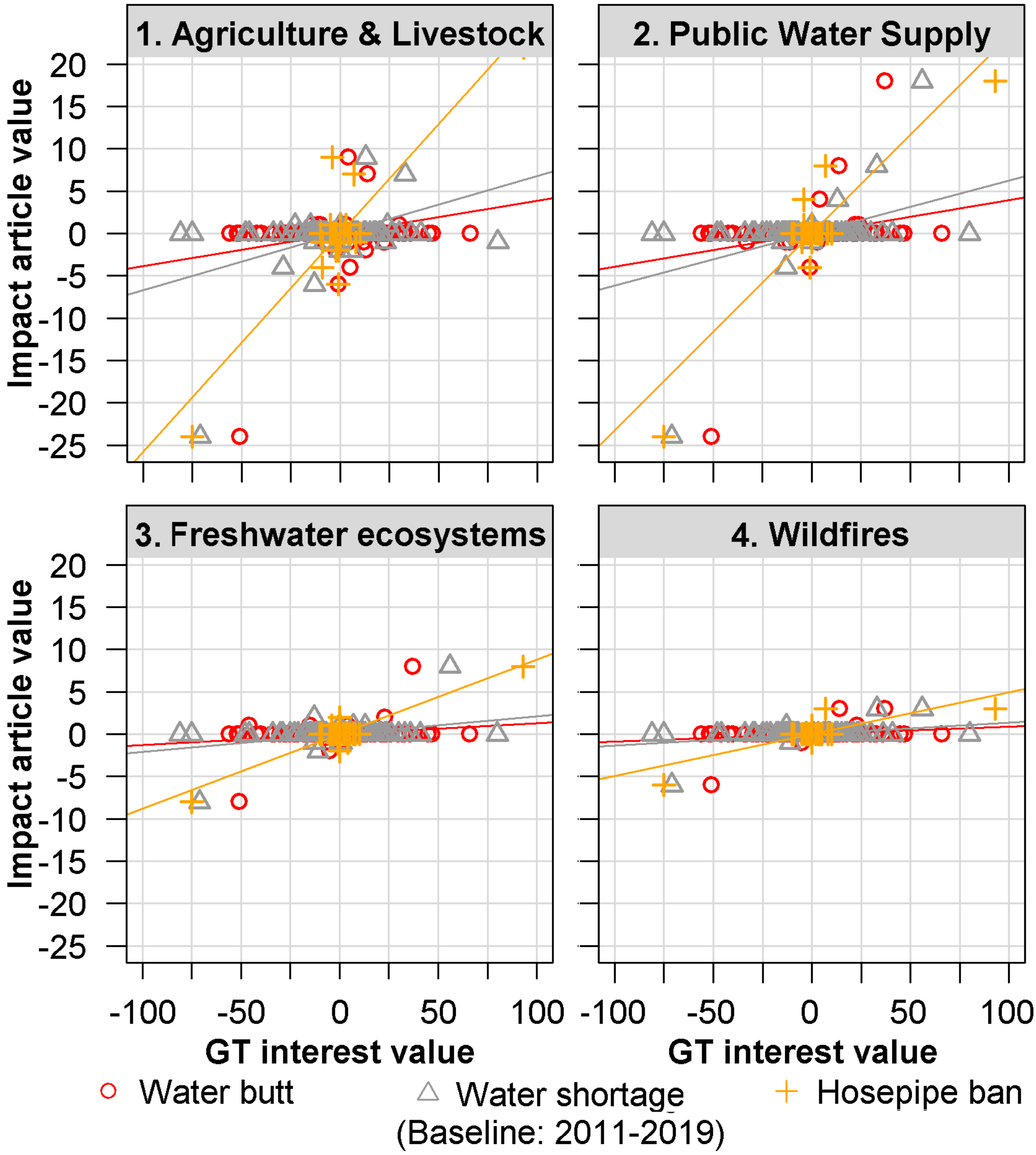


3.2.3. Droughts in Ireland

In Ireland, the terms “hosepipe ban”, “water shortage”, and “water butt” were most strongly linked to the number of newspaper reports about drought impacts on agriculture/livestock, PWS, freshwater/terrestrial, soils, and wild fires (Figure 8). As before, these associations tend to be dominated by a small number of outliers. Nonetheless, GT terms such as “hosepipe ban” are a good proxy for high historical interest by (print) media in agricultural/ livestock and PWS. Presumably, this interest is being driven by a common factor – severe drought (notably in summer 2018).

Two other points emerge from this analysis. First, the correlation between the GT data for "hosepipe ban" and each of the IDID categories was strong (r > 0.7) relative to other terms – as evidenced by the steeper gradients for this term in Figure 8. Second, agriculture and livestock, and PWS are prominent because of the greater number of impact reports within the period 2011-2019 when compared with all other categories (which had less than 10 articles each in the same period).

Figure 8 Associations between GT interest in "water butt", "water shortage", and "hosepipe ban" versus the number of drought impact articles (for IDID categories with >10 articles) in Ireland during the period 2011 to 2019. Note that impact article counts and GT interest values were detrended.



4. Discussion

This preliminary analysis shows the potential for relating GT search interest around drought terms with quantitative indicators of water availability and impacts – here demonstrated using available national precipitation and outflow series for England and counts of newspaper reports of selected drought impacts in Ireland.

The GT data since year 2011 reveal upward trends in search interest around several themes, most notably for “grey water”, “water leak”, “water meter”, and “water use”. This suggests that searches have been focused more on outdoor water saving rather than indoor measures (as the interest volumes decrease from water butt > watering can > grey water > dual flush toilet/ tap aerator > water saving shower head). The interest in some search terms (“water butt” and “irrigation system”) are also weakly but significantly correlated with national precipitation and runoff series for England. For Ireland, there were no significant correlations between GT search terms and total monthly precipitation values, derived from the recently updated Island of Ireland Precipitation series (Murphy et al., 2018; Noone et al., 2016).

Insights from this analysis that are potentially relevant to water authorities, policy-makers, and water companies are:

- Rising search interest in water saving technologies suggests a growing awareness of household-level solutions to water scarcity during droughts;
- Greater search interest around outdoor water saving measures than indoor measures may signal opportunities for more targeted public information campaigns;
- Search interest in Defra and the Environment Agency is more associated with floods than droughts, hence public relations teams might consider ways of improving awareness of roles played by these organisations during droughts;
- Search interest in water companies in England is more likely to be associated with hosepipe bans than water leakage (although this varies by company).

These findings could also lead to the development of novel indicators to inform drought management and water resource planning. For instances, real-time GT data on searches for outdoor water saving and use (i.e., water butts and cans, grey water) plus irrigation systems might track the rising severity of a meteorological drought. Our evidence from Ireland suggests that these terms could also reflect real-time impacts on agriculture, livestock and PWS, as well as harm to natural freshwater and terrestrial environments. Rising volumes of searches about water leakage and metering, paired with water use and shortages, may signal greater public awareness of the links between these issues and hence scope for policy interventions. This could be a manifestation of evolving notions of water consumer rights and responsibilities (Taylor et al., 2009). Variations in the phasing of relative interest in drought and heatwave may also capture shifting societal concerns during an event –signatures that are unique to each heat-drought episode. Water companies might note that searches for their names (principally Thames Water, Southern Water, Anglian Water, Severn Trent Water, Yorkshire Water) frequently associate with interest in hosepipe bans and meters.

GT data appear to cover most of the desirable attributes for an indicator given in Table 3. Search volumes are benchmarked (individually or relative to other terms); sensitive to hydroclimatic conditions (as evidenced by correlations with rainfall and runoff); based on free and open data that are updated in real-time; reflective of changes in public interest and awareness; and yield insights that are decision-relevant (around messaging, marketing, and water consumer behaviour). However, although GT data are available from 2004, there have been several unspecified system changes since then (most recently on 1 January 2022) so the homogeneity of GT output is uncertain. Information seeking behaviour is shaped by many factors, such as prior knowledge and characteristics of the subject matter or product (Jun et al., 2018). Hence, there a raft of factors that potentially confound interpretations. On this basis, GT data would score ~7/10 against the criteria listed in Table 3.

Our initial findings and suggestions should be treated with caution because of other specific shortcomings of GT data. For example, particular search terms can have multiple meanings: “runoff” cannot be included because of links to electoral runoffs (in the US State of Georgia, and Senate); and “showering” is biased by searches for Nic Showering (one of The Apprentice 2022 contestants). Some data are even available for misspelt terms such as “drout”! The term “water cycle” has strong intra-annual variations that might be attributable to school coursework and examinations. As might be expected there are relatively few searches and hence robust data for technical terms like river pollution, water saving, water level, and public water supply. The geographical domain reflects the location of the searcher rather than the subject matter. Finally, it must be kept in mind that public interest in a drought-related topic –such as water saving devices for the home or leakage from the water distribution network – does not necessarily translate into changes in behaviour or expenditure.

Despite these reservations our analysis shows the potential for GT data to inform drought management and water planning. The UK still lacks a one-stop-shop, public domain, environmental indicator set because the need to make an economic case remains a persistent obstacle. Nonetheless, techniques are being developed for valuing climate services, including when building climate resilience (Watkiss et al., 2021). Some assert that resilience metrics and indicators are important tools for tracking risks and climate actions (Wilby, 2020). Others claim that the things we really care about are not always quantifiable: Measurement requires stopping the action, getting outside of it and holding it up against a yardstick, exactly the opposite of the activity that would create products or ship them, make customers happy or move our business forward in any way (Ryan, 2014). However, we assert that the economy, immediacy and intimacy of GT makes this a powerful tool worthy of further consideration for environmental applications, including water management.

5. Conclusions

This paper provides an overview of indicators used by the UK water sector to reduce disruption to supplies and harm to the environment. Drought-related risks are also woven into many national climate change indicators. However, there remain major gaps in national capability to track impacts of droughts (in near real-time) as well as for monitoring progress in building resilience to future drought risks. Considerations of cost, immediacy, access, consistency, relevance, reliability, and others, have to be addressed when evaluating the suitability of information for indicators. Here, we show the largely untapped potential of Google Trends (GT) data and newspaper archives as resources for developing novel drought indicators.

Our preliminary analysis for England and Ireland shows that GT search interest can track the temporal evolution of a drought, along with changes in public awareness and enquiries into drought impacts and adaptation measures. Search volumes for terms such as “water butt” and “hosepipe ban” are significantly correlated with conventional hydroclimatic data. There is also evidence of longer-term growth of interest in water saving technologies and techniques, especially around outdoor water use. However, we are mindful that many factors influence Google searches, not least the prior knowledge of the user, and that GT has limitations. Nonetheless, the data yield insights about associations in search terms such as “Environment Agency” and “flood” (as opposed to “drought”), or related queries about water companies and hosepipe bans. Such knowledge could be used to shape public information campaigns and policy development. The GT analysis for Ireland could also be replicated using drought impact reports within existing newspapers archives for England. Another possibility worthy of exploration is the potential use of novel indicators for forecasting drought impacts.

References

ADAS, 2019. Research to update the evidence base for indicators of climate-related risks and actions in England [accessed 23 December 2022].

ADAS, 2021. Research to review and update indicators of climate-related risks and actions in England [accessed 23 December 2022].

Antwi, S.H., Rolston, A., Linnane, S. and Getty, D. 2022. Communicating water availability to improve awareness and implementation of water conservation: A study of the 2018 and 2020 drought events in the Republic of Ireland. Science of The Total Environment, 807, 150865.

Arnell, N.W., Freeman, A., Kay, A.L., Rudd, A.C. and Lowe, J.A. 2021. Indicators of climate risk in the UK at different levels of warming. Environmental Research Communications, 3, 095005.

Bachmair, S., Stahl, K., Collins, K., Hannaford, J., Acreman, M., Svoboda, M., Knutson, C., Smith, K.H., Wall, N., Fuchs, B. and Crossman, N.D. 2016b. Drought indicators revisited: the need for a wider consideration of environment and society. Wiley Interdisciplinary Reviews: Water, 3, 516-536.

Bachmair, S., Svensson, C., Hannaford, J., Barker, L.J. and Stahl, K. 2016a. A quantitative analysis to objectively appraise drought indicators and model drought impacts. Hydrology and Earth System Sciences, 20, 2589-2609.

Brázdil, R., Kiss, A., Luterbacher, J., Nash, D.J. and Řezníčková, L. 2018. Documentary data and the study of past droughts: a global state of the art. Climate of the Past, 14, 1915-1960.

Climate Change Committee, 2021c. Progress in adapting to climate change 2021 Report to Parliament. https://www.theccc.org.uk/wp-content/uploads/2021/06/Progress-in-adapting-to-climate-change-2021-Report-to-Parliament.pdf [accessed 22 December 2022]

Climate Change Committee, 2021a. Evidence for the third UK Climate Change Risk Assessment (CCRA3). Summary for England [accessed 23 December 2022].

Climate Change Committee, 2021b. Progress Report to Parliament [accessed 23 December 2022].

Dayrell, C., Svensson, C., Hannaford, J., McEnery, T., Barker, L.J., Baker, H. and Tanguy, M. 2022. Representation of drought events in the United Kingdom: Contrasting 200 years of news texts and rainfall records. Frontiers in Environmental Science, 10, 760147.

de Brito, M.M., Kuhlicke, C. and Marx, A. 2020. Near-real-time drought impact assessment: a text mining approach on the 2018/19 drought in Germany. Environmental Research Letters, 15, 1040a9.

Defra, 2020. Outcome Indicator Framework for the 25 Year Environment Plan: 2020 Update [accessed 23 December 2022].

Department of the Environment, Transport and the Regions (DETR), 1999. Indicators of Climate Change in the UK. Climatic Research Unit, University of East Anglia and the Centre for Ecology & Hydrology, Wallingford, UK.

Ekström, M., Gutmann, E.D., Wilby, R.L., Tye, M.R. and Kirono, D.G. 2018. Robustness of hydroclimate metrics for climate change impact research. Wiley Interdisciplinary Reviews: Water, 5, 1288.

Environment Agency (2020). Meeting our future water needs: a national framework for water resources [accessed 23 December 2022].

Environment Agency, 2022. Water and Sewerage Companies in England: EPA Metric Guide for 2021 [accessed 23 December 2022].

Hannaford, J., Collins, K., Haines, S. and Barker, L.J., 2019. Enhancing drought monitoring and early warning for the United Kingdom through stakeholder coinquiries. Weather, Climate, and Society, 11, 49-63.

Jobbová, E., Crampsie, A., Murphy, C., Ludlow, F., McLeman, R.A., Horvath, C., Seifert, N., Myslinski, T. and Sente, L. 2023. The Irish Drought Impacts Database (IDID): A 287-year database of drought impacts derived from newspaper archives. Geosciences Data Journal, submitted.

Jun, S.P., Yoo, H.S. and Choi, S. 2018. Ten years of research change using Google Trends: From the perspective of big data utilizations and applications. Technological Forecasting and Social Change, 130, 69-87.

Kam, J., Stowers, K. and Kim, S. 2019. Monitoring of drought awareness from Google Trends: a case study of the 2011–17 California drought. Weather, Climate, and Society, 11, 419-429.

Kim, S., Shao, W. and Kam, J. 2019. Spatiotemporal patterns of US drought awareness. Palgrave Communications, 5, 107.

Lee, J.W., Hong, E.M., Kim, J.U., Jang, W.J., Jung, C.G. and Kim, S.J. 2022. Evaluation of agricultural drought in South Korea using socio-economic drought information. International Journal of Disaster Risk Reduction, 74, p.102936.

Mavragani, A., Sypsa, K., Sampri, A. and Tsagarakis, K.P. 2016. Quantifying the UK online interest in substances of the EU watchlist for water monitoring: diclofenac, estradiol, and the macrolide antibiotics. Water, 8, 542.

Murphy, C., Noone, S., Duffy, C., Broderick, C., Matthews, T. and Wilby, R.L. 2017. Irish droughts in newspaper archives: rediscovering forgotten hazards? Weather, 72, 151-155.

Murphy, C., Broderick, C., Burt, T.P., Curley, M., Duffy, C., Hall, J., Harrigan, S., Matthews, T.K.R., Macdonald, N., McCarthy, G., McCarthy, M.P., Mullan, D., Noone, S., Osborn, T.J., Ryan, C., Sweeney, J., Thorne, P.W., Walsh, S., and Wilby, R.L. 2018. A 305-year continuous monthly rainfall series for the Island of Ireland (1711–2016). Climate of the Past, 14, 413-440.

Murphy, C., Wilby, R.L., Matthews, T., Horvath, C., Crampsie, A., Ludlow, F., Noone, S., Brannigan, J., Hannaford, J., McLeman, R. and Jobbova, E. 2020. The forgotten drought of 1765–1768: Reconstructing and re-evaluating historical droughts in the British and Irish Isles. International Journal of Climatology, 40, 5329-5351.

Noone, S., Murphy, C., Coll, J., Matthews, T., Mullan, D., Wilby, R.L. and Walsh, S. 2016. Homogenisation and analysis of an expanded monthly rainfall network for the Island of Ireland (1850-2010). International Journal of Climatology, 36, 2837-2853.

Noone, S., Broderick, C., Duffy, C., Matthews, T., Wilby, R.L. and Murphy, C. 2017. A 250-year drought catalogue for the island of Ireland (1765–2015). International Journal of Climatology, 37, 239-254.

O'Connor, P., Murphy, C., Matthews, T. and Wilby, R.L. 2022a. Historical droughts in Irish catchments 1767–2016. International Journal of Climatology, 42, 5442-5466.

O'Connor, P., Murphy, C., Matthews, T. and Wilby, R.L. 2022b. Relating drought indices to impacts reported in newspaper articles. International Journal of Climatology, https://doi.org/10.1002/joc.7946

Park, C.K., Kam, J., Byun, H.R. and Kim, D.W. 2022. A self-calibrating effective drought index (scEDI): evaluation against social drought impact records over the Korean Peninsula (1777–2020). Journal of Hydrology, 613, 128357.

Paterson, S.K. and Godsmark, C.N. 2020. Heat-health vulnerability in temperate climates: lessons and response options from Ireland. Globalization and Health, 16, 29.

Ryan, L. 2014. 'If You Can't Measure It, You Can't Manage It': Not True. Forbes [accessed 23 December 2022].

Stahl, K., Blauhut, V., Kohn, I., Ac acio, V., Assimacopoulos, D., Bifulco, C., De Stefano, L., Dias, S., Eilertz, D., Freilingsdorf, B., Hegdahl, T.J., Kampragou, E., Kourentzis, E., Melsen, L., van Lanen, H.A.J., van Loon, A.F., Massarutto, A., Musolino, D., De Paoli, L., Senn, L., Stagge, J.H., Tallaksen, L.M. and Urquijo, J. 2012. A European drought impact report inventory (EDII): design and test for selected recent droughts in Europe. Wageningen: Wageningen Universiteit. DROUGHT-R&SPI technical report: 3.

Taylor, V., Chappells, H., Medd, W. and Trentmann, F. 2009. Drought is normal: the socio-technical evolution of drought and water demand in England and Wales, 1893–2006. Journal of Historical Geography, 35, 568-591.

Thompson, J.J., Wilby, R.L., Matthews, T. and Murphy, C. 2022. The utility of Google Trends as a tool for evaluating flooding in data-scarce places. Area, 54, 203-212.

van der Schrier, G., Allan, R.P., Ossó, A., Sousa, P.M., Van de Vyver, H., Van Schaeybroeck, B., Coscarelli, R., Pasqua, A.A., Petrucci, O., Curley, M. and Mietus, M. 2021. The 1921 European drought: impacts, reconstruction and drivers. Climate of the Past, 17, 2201-2221.

Watkiss, P. and Hunt, A. 2021. Methods and case studies for valuing climate services in the UK Climate Resilience Programme. EMS Annual Meeting 2021, online, 6–10 Sep 2021, EMS2021-119, https://doi.org/10.5194/ems2021-119.

Wilby, R.L. 2020. Resilience viewed through the lens of climate change and water management. Water, 12, 2510.

Wilby, R.L. and Johnson, M.F. 2020. Climate variability and implications for keeping rivers cool in England. Climate Risk Management, 30, 100259.

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