Exposure and adverse effects of chemicals on wildlife in the environment: interim H4 indicator

Supporting information and data

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Version: 1.0

We are the Environment Agency. We protect and improve the environment.

We help people and wildlife adapt to climate change and reduce its impacts, including flooding, drought, sea level rise and coastal erosion.

We improve the quality of our water, land and air by tackling pollution. We work with businesses to help them comply with environmental regulations. A healthy and diverse environment enhances people's lives and contributes to economic growth.

We cannot do this alone. We work as part of the Defra group (Department for Environment, Food and Rural Affairs), with the rest of government, local councils, businesses, civil society groups and local communities to create a better place for people and wildlife.

Executive summary

The H4 indicator 'Exposure and adverse effects of chemicals on wildlife in the environment' is one of a suite of indicators in the 25-Year Environment Plan (25-YEP) Outcome Indicator Framework. The indicator contributes to measuring whether we are moving towards the goal of 'managing exposure to chemicals' within the 25-YEP. [Available updates to indicators under the framework](#) are reported annually on gov.uk alongside reports outlining environmental improvements made through the 25-YEP.

This current report covers progress on the development of the H4 indicator and shows our interim version, which uses a dashboard approach, and the corresponding data analysis behind it. The indicator is based on chemical concentrations found in water and in different organisms – sparrowhawk/red kite, red fox, freshwater fish, otter, blue mussel, dab, and harbour porpoise. It covers 3 environmental compartments: terrestrial, freshwater and marine (estuarine, coastal and offshore). The chemicals presented are representative of 3 groups requiring priority management as highlighted in the 25-YEP: persistent, bioaccumulative and toxic (PBT) substances, heavy metals, and pesticides/biocides.

The dashboard illustrates statistically significant trends over time in environmental concentrations for the presented chemicals (see Figure). Available year ranges for assessing trends vary for the different data sources. We have considered data up to 2019 where possible, although some datasets represented in the dashboard stop at earlier years within the last decade. The indicator also considers potential risks to wildlife from chemicals by comparing the most-recent year(s) concentration data against relevant environmental protection thresholds, if available. This assessment of risk provides a surrogate for effects reporting for this interim indicator. For PBT substances, the thresholds may differ from those used under other reporting regimes which are based on the protection of humans as the most-sensitive receptor.

Overall, only a limited number of datasets show statistically significant changes in chemical concentrations over time. This may be a consequence of some chemicals, such as PBT substances, being slow to respond to change. It may also reflect that the data are for a period up to the beginning of the 25-YEP timeline; therefore, some management actions may be in their early stages. Exceedance of thresholds across sites or in individuals is seen for all 3 chemical groups, which is not unexpected given the choice of these substances as likely or known substances of concern.

For PBT substances, downward trends are observed for polybrominated diphenyl ethers (PBDEs) and polychlorinated biphenyls (PCBs) in marine fish (common dab) and for PBDEs and perfluorooctanesulfonic acid (PFOS) in harbour porpoise. These trends are particularly evident for PBDEs.

The proportion of sites or samples exceeding thresholds is low for PBDEs compared with those for the other PBT substances assessed. Exceedance of thresholds is greatest for mercury in the freshwater and marine environments, followed by PCBs in the marine

environment. The result for mercury in marine fish, however, was based on a threshold that could be considered over-precautionary for the tissue examined.

For heavy metals, downward trends have been observed for nickel and zinc in sparrowhawks, though the data are only available up to 2014. There is an upward trend for nickel in dab, which is driven by eastern and southern coastal marine sites.

Water concentration data for lead, cadmium, nickel, copper, and zinc exceed thresholds at between 1 and 24% of freshwater monitoring sites. This level of exceedance is also seen for nickel and zinc in estuarine and coastal waters, although the nickel result is only driven by one site. Zinc shows the highest rate of threshold exceedance of the metals in both freshwater and estuarine and coastal waters.

While the freshwater data for heavy metals overall show no statistically significant change in concentrations from 2014 to 2019, these results can be split into two types: those for waters affected by 'abandoned metal mines' and those for sites in 'other' locations. Cadmium and copper concentrations show downward trends for the 'other' sites over the assessed time period. No other trends in concentrations over time are seen at either type of site. The elevated levels of most metals in waters affected by abandoned metal mines mean that these sites constitute the majority of those overall exceeding risk thresholds. For nickel, 'other' locations comprise the majority of those overall at risk.

It is not possible to assess trends currently for pesticides and second-generation anticoagulant rodenticides (SGARs). Threshold exceedance is indicated for less than a quarter of sites or individuals considered for a broad range of pesticides in freshwater and SGARs in red kite.

There is some variability across the different assessments in the indicator in terms of years assessed, congeners reviewed for PBDEs and PCBs, and in the basis of the thresholds used. Our aim has been to make the assessment comprehensive and consistent as possible using readily available data.

An independent review of this reporting approach was conducted by expert committees. This supported the dashboard approach as an exposure indicator, but emphasised the need to cover effects information to illustrate impacts on wildlife from chemicals in the environment.

We are continuing to develop this indicator guided by recommendations from an initial trial of the dashboard approach in 2020, comments from the independent review of that work, and improvements realised through constructing the presented interim version. Progress on the indicator will be reported through future Outcome Indicator Framework annual reports. We will seek to address data gaps for all substances to get a fuller picture across compartments and improve our ability to report exposure trends. Future work will also consider methods for reporting the effects on wildlife of chemicals in the environment.

The H4 indicator is a collaborative piece of work steered by representatives from Defra, the Environment Agency, the Health and Safety Executive, Natural England, and UKCEH. Contributors include these and other organisations such as Cefas and Fera.

Figure Exposure of wildlife to chemicals in the environment - the interim H4 indicator dashboard

		TERRESTRIAL		FRESHWATER			MARINE			
		Sparrowhawk / red kite	Red fox	Freshwater	Roach / chub / brown trout	Otter	Estuarine / coastal waters	Blue mussels	Dab	Harbour porpoise
PBT substances	Mercury	below threshold (no change)		NR	>75% (no change)	below threshold (no change)	NR	>75%	>75%	
	PBDEs			NR	below threshold		NR	below threshold	1–24% (decreasing)	below threshold (decreasing)
	PCBs			NR	no threshold (no change)		NR	50–74%	25–49%	50–74% (no change)
	PFOS			1–24%	1–24%		NR		below threshold	below threshold (decreasing)
Heavy metals	Lead	below threshold (no change)		1–24% (no change)			below threshold (no change)	no threshold (no change)	no threshold (no change)	
	Cadmium	below threshold (no change)		1–24% (no change)		no threshold (no change)	below threshold (no change)	no threshold (no change)	no threshold (no change)	
	Nickel	no threshold (decreasing)		1–24% (no change)			1–24% (no change)	no threshold (no change)	no threshold (increasing)	
	Copper	no threshold (no change)		1–24% (no change)			below threshold (no change)	no threshold (no change)	no threshold (no change)	
	Zinc	no threshold (decreasing)		1–24% (no change)			1–24% (no change)	no threshold (no change)	no threshold (no change)	
Pesticides and biocides	Pesticides			1–24%			NR	NR	NR	NR
	SGARs	1–24%					NR	NR	NR	NR

Key

Data sources: Sparrowhawk / red kite; Red fox; Freshwater; Roach / chub / brown trout; Otter; Estuarine / coastal waters; Blue mussels; Dab; Harbour porpoise

Acronyms: PBT: persistent, bioaccumulative and toxic; PBDEs: polybrominated diphenyl ethers; PCBs: polychlorinated biphenyls; PFOS: perfluorooctanesulfonic acid; SGARs: second-generation anticoagulant rodenticides; NR: not relevant – unlikely to be an exposure route for that substance

Trend: Increasing concentrations (↑); No observed change in concentrations (↔); Decreasing concentrations (↓)

Only statistically significant trends in environmental concentrations are shown for upward and downward arrows; no arrow indicates minimum requirements for trend assessment are not met. Available year ranges for assessing trends vary and trends are only assessed for data sources with at least 5 full years of change (6 independent sampling years).

Risk:
- More than 75% sites/samples above threshold
- 50 to 74% sites/samples above threshold
- 25 to 49% sites/samples above threshold
- 1 to 24% sites/samples above threshold
- All sites/individuals or population average below threshold
- No threshold available; not currently able to assess risk

Assessment is based on comparison of concentration data for the most-recent year, 2 years for dab and 3 years for PFOS and heavy metals in water.

Notes:
1. Blank spaces indicate there are currently insufficient or no comparable data available to allow trend or risk reporting.
2. Data cover up to and including 2019 where available; exceptions are mercury (2013) and heavy metals (2014) in sparrowhawk, mercury and cadmium in otter (2016), and PBT substances in harbour porpoise (2018).

Contents

1 Introduction

The UK Government's 25-Year Environment Plan (25-YEP) was published in 2018 ([UK Government, 2018](#)).

One of the goals within the 25-YEP is 'managing exposure to chemicals'. This goal covers the safe use and management of chemicals and intends to ensure that levels of harmful chemicals entering the environment are significantly reduced. It is supported by a range of actions and specific targets relating to decreasing chemical emissions ([UK Government, 2018](#)). To demonstrate any effect of those measures and progress towards the goal, it is important to consider other environmental information in relation to chemicals alongside emissions information.

To assist the monitoring of progress against commitments in the 25-YEP, the Department for Environment, Food and Rural Affairs (Defra) established a suite of indicators under an Outcome Indicator Framework ([Defra, 2019](#)). Within that suite is the H4 indicator 'Exposure and adverse effects of chemicals on wildlife in the environment'.

The aim of the H4 indicator is:

- To show how exposure of wildlife on land or in water to harmful chemicals is changing, and
- To see whether wildlife is impacted by environmental exposure to such chemicals and compare this over time.

Other indicators under the Outcome Indicator Framework provide information complementary to H4 and support the measurement of progress towards the 25-YEP goal. Directly related is the indicator H3 'Emissions of mercury and persistent organic pollutants to the environment'. Additionally, some of the water indicators such as B1 'Pollution loads entering water' help describe pressures on the environment relating to chemicals. The Outcome Indicator Framework is updated yearly alongside the 25-YEP annual report ([Defra, 2020](#)).

Section 2 of this report describes progress on the development of the H4 indicator. Section 3 explains the approach taken for reporting and provides the interim indicator as a dashboard along with a summary. Section 4 of the report describes the underlying data and its translation into the indicator. Tables and figures have been numbered based on their corresponding first-level subsection to ensure the text and data are easily associated. For example, the first table in Section 4.1 is numbered Table 4.1.1, the second Table 4.1.2, etc.

The interim version of the H4 indicator tracks changes over time in the exposure of wildlife to chemicals. It does this through looking at measured chemical concentrations in the environment in which wildlife live and within specific species. It covers 3 environmental compartments: terrestrial, freshwater and marine (estuarine, coastal and offshore). Chemical groups considered are persistent, bioaccumulative and toxic (PBT) substances,

heavy metals, and pesticides and biocides. The 3 groups reflect chemicals highlighted for management action in the 25-YEP.

The indicator focusses on readily available data for these groups that are generated by repeat monitoring or have the potential to become regular sources of information. Much of the monitoring currently in place is undertaken for reporting commitments, such as:

- The Convention for the Protection of the Marine Environment of the North-East Atlantic (the 'OSPAR Convention') (OSPAR Commission, 2020),
- The Marine Strategy Regulations 2010 (UK Government, 2010) and the assessment of Good Environmental Status in Regional Seas,
- The Water Environment (Water Framework Directive) Regulations 2017 (UK Government, 2017), and
- The Water Framework Directive (Standards and Classification) Directions (England and Wales) 2015 (UK Government, 2015).

In addition, the indicator draws on data available from other monitoring campaigns and research activities.

The interim indicator also assesses potential risk to wildlife from chemicals by comparing the available monitoring data against relevant environmental protection thresholds where possible. This helps add context to the levels of chemicals reported in the environment and acts as a surrogate for effects data while the reporting of that aspect of the indicator is still undergoing exploration.

Thresholds are generally derived from effects data either generated in the laboratories or based on field observations. Effects can be measured in a variety of direct ways in terms of impairment, such as changes in the morphology of an individual, reproductive success, and death. Indirect examples include changes in population levels or food resource over time.

As part of the H4 indicator development, we are considering a variety of indicators that could answer questions on indirect and sublethal exposure and effects of chemicals. Those in early development stages have not been described in this report.

We are continuing to develop this indicator guided by recommendations from the initial trial of the dashboard approach (Shore et al. 2020, Shore and Walker, 2020), comments from the independent review of that work, the UKCEH/Natural England Pesticide Monitoring Review, and improvements realised through constructing the presented interim version.

2 Development of the H4 indicator

2.1 Overview of the work during 2020/2021

The H4 indicator is a multi-organisational piece of work that is informed through technical and policy stakeholders on a steering group as well as individual experts on various related working groups (Figure 2.1.1).

Figure 2.1.1 Structure of the working groups established to support the development and reporting of the H4 indicator
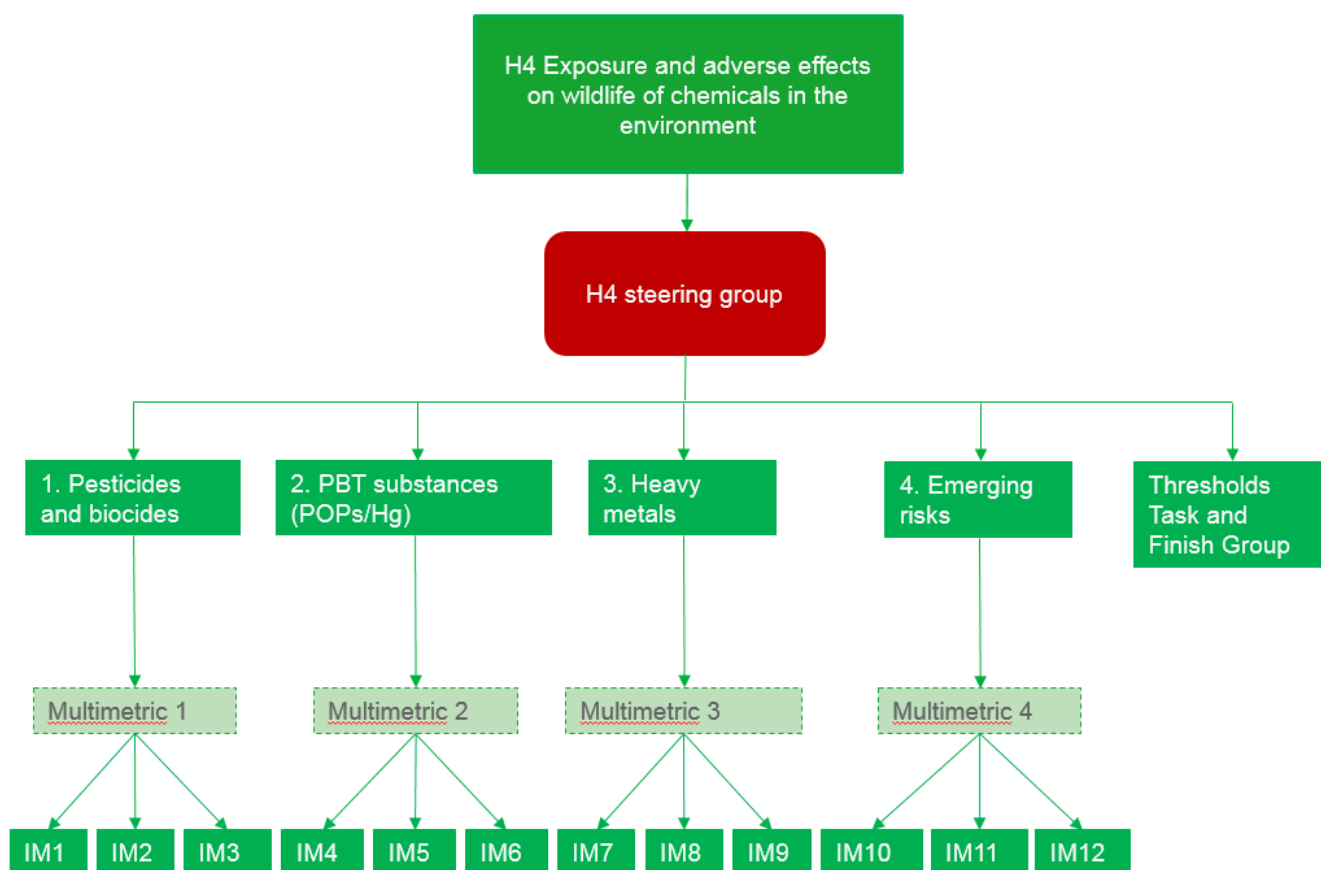
IM = individual metrics

The development of the H4 indicator up until the end of March 2020 is covered in Shore et al. (2020). That publication described the choice of the specific contaminant classes under H4, the identified potential datasets and the selection of the dashboard concept for reporting. Importantly, Shore et al. (2020) trialled the dashboard approach and identified issues for its further development (Shore and Walker, 2020).

As a result of that process we have:

- Sought independent review of our indicator (Section 2.2).
- Refined our reporting to include additional available data on concentrations of perfluorooctanesulfonic acid (PFOS), polychlorinated biphenyls (PCBs) and metals

as proposed by the working groups (Section 2.3). We have also incorporated data for PBTs and heavy metals in blue mussels (*Mytilus edulis*) into our indicator.

- Expanded our working groups to include a thresholds task and finish (T&F) group (Section 2.3).
- Initiated a desk-top study to look for further suitable thresholds (Section 2.3).
- Initiated work assessing the reliability of the data – 'power of the metrics' – to show change over time (Section 2.2).
- Maintained an awareness of or linked in with other related areas of work, such as the development of parallel indicators involving chemicals or ongoing exposure and effects research work.

The interim indicator is being published as an Experimental Statistic in order to facilitate user involvement in the development of this indicator. We would therefore welcome any feedback on these statistics, particularly on their usefulness and value, via 25YEPindicators@defra.gov.uk.

2.2 Independent review

In July 2019, we took the H4 indicator work to the Hazardous Substances Advisory Committee (HSAC) and the Expert Committee on Pesticides (ECP) for comment (HSAC, 2020; ECP, 2020). The comments received were supportive of the dashboard as an exposure indicator, but highlighted the need to address data gaps, investigate the power of the metrics included to show change, and develop the biological effects side of the indicator.

The pragmatic approach used for pesticides (see Section 4.23) was supported with future alterations to consider risk from chronic exposure rather than acute. The method considers the combined risk to the freshwater environment owing to over 300 pesticides. However, the reviewers suggested that the indicator could go further to assess the integrated impacts of diverse suites of pollutants in combination with or in addition to those from pesticides. Further methods for directly assessing pesticide effects in the freshwater environment are currently being researched by the Environment Agency.

Both committees emphasised the importance of ensuring the work reached the appropriate audiences and that findings were fed into policy development. We have policy partners involved in the work and through our publication of the interim version will continue to take on board feedback to ensure the indicator is useful (see Section 3.1).

Since the independent review, work has started looking at the power of an initial group of datasets within the indicator to show change over time. This is an important area of work to ensure we will be able to identify trends in environmental concentrations over the lifetime of the 25-YEP and to help inform future monitoring. We are seeking to address data gaps for all substances to get a fuller picture across compartments and improve our ability to report exposure trends. Future work will also consider methods for reporting the effects on wildlife of chemicals in the environment.

In response to some of the review comments, our dashboard presentation now distinguishes more clearly between trend and risk information (Section 3).

2.3 Working groups

Since the trial of the indicator by UKCEH (Shore et al., 2020; Shore and Walker, 2020), the working groups for the different chemical types (Figure 2.1.1) have assessed the availability of other data sources and considered the possible inclusion of additional substances. The groups have also contributed to the provision of the current H4 indicator data and its quality assurance.

No readily available additional types of exposure data were identified for reporting. For the existing data sources, the PBT group identified data for concentrations of PCBs in freshwater fish and PFOS in freshwater to improve the representation of these chemicals across the indicator. The heavy metals group confirmed the inclusion of nickel, copper and zinc across all compartments where possible. These substances were identified earlier in the indicator's development, but were not included in the initial trial. They were selected because of their inclusion under existing monitoring programmes as pollutants of concern.

Based on the comments from Shore and Walker (2020), we also established a thresholds T&F group. The purpose of this group is to review the appropriateness of the threshold values used in the reporting of the H4 indicator. The group also considers suitable methods for assessment to ensure consistency within the indicator and with other reporting regimes, as far as reasonably possible.

The recommendations from the T&F group on the thresholds and their application have been incorporated into the data assessments in Section 4. The group have proposed additional values to be used, where available, for the introduced substances mentioned above and for blue mussels as a newly introduced data source to the indicator.

A desk-top study is underway to look for further suitable thresholds in the primary and grey literature. It will determine the availability of additional thresholds for a small number of the contaminants and receptors of interest, particularly for the terrestrial environment where more gaps in relevant information exist.

Further overview on the thresholds used for generating the interim indicator can be found in Section 3.

To date, the active working groups have been those representing the first 3 chemical groups and the thresholds T&F group (Figure 2.1.1). A fourth working group on emerging risks was proposed at the start of the H4 indicator work (Shore et al., 2020). We plan to establish this group in the near future.

3 Indicator dashboard and reporting summary

3.1 Overview of the dashboard construction

The interim H4 indicator dashboard is shown in Figure 3.1.1.

The information in the interim indicator reflects our assessment of readily available data for which we can explore changes in chemical concentrations over time and assess risk, as a surrogate for effects, using available thresholds. It is based on data up until the end of 2019, although the data sources differ in what time periods are available (see Section 3.1.1).

In the dashboard, columns represent environmental compartments and available data for matrices within them. Rows are clustered by the 3 chemical groups considered under the indicator with representative priority chemicals shown.

Where entries are given as 'NR', this is due to the matrix being less relevant as an exposure route. For example, most PBT substances are unlikely to be detected in water as they are highly insoluble and more likely to be found in organisms such as fish. Pesticides are less of an issue in the marine than the freshwater compartment because of their greater distance from emission sources and likelihood of them being degraded or dispersed to negligible levels.

Some datasets are reported within the dashboard as blanks. This is used for cases for which there are either:

- No available data via that data source, or
- Available data, but not enough to determine a trend over time and there are no thresholds to allow a comparison with the analysed concentrations.

Figure 3.1.1 Exposure of wildlife to chemicals in the environment - the interim H4 indicator dashboard

		TERRESTRIAL		FRESHWATER			MARINE			
		Sparrowhawk / red kite	Red fox	Freshwater	Roach / chub / brown trout	Otter	Estuarine / coastal waters	Blue mussels	Dab	Harbour porpoise
PBT substances	Mercury			NR			NR			
	PBDEs			NR			NR			
	PCBs			NR			NR			
	PFOS							NR		
Heavy metals	Lead									
	Cadmium									
	Nickel									
	Copper									
	Zinc									
Pesticides and biocides	Pesticides						NR	NR	NR	NR
	SGARs						NR	NR	NR	NR

Key

Data sources: Sparrowhawk / red kite; Red fox; Freshwater; Roach / chub / brown trout; Otter; Estuarine / coastal waters; Blue mussels; Dab; Harbour porpoise

Acronyms: PBT: persistent, bioaccumulative and toxic; PBDEs: polybrominated diphenyl ethers; PCBs: polychlorinated biphenyls; PFOS: perfluorooctanesulfonic acid; SGARs: second-generation anticoagulant rodenticides; NR: not relevant – unlikely to be an exposure route for that substance

Trend:
- ↑ Increasing concentrations
- ↔ No observed change in concentrations
- ↓ Decreasing concentrations

Only statistically significant trends in environmental concentrations are shown for upward and downward arrows; no arrow indicates minimum requirements for trend assessment are not met. Available year ranges for assessing trends vary and trends are only assessed for data sources with at least 5 full years of change (6 independent sampling years).

Risk:
- More than 75% sites/samples above threshold
- 50 to 74% sites/samples above threshold
- 25 to 49% sites/samples above threshold
- 1 to 24% sites/samples above threshold
- All sites/individuals or population average below threshold
- No threshold available; not currently able to assess risk

Assessment is based on comparison of concentration data for the most-recent year, 2 years for dab and 3 years for PFOS and heavy metals in water.

Notes:
1. Blank spaces indicate there are currently insufficient or no comparable data available to allow trend or risk reporting.
2. Data cover up to and including 2019 where available; exceptions are mercury (2013) and heavy metals (2014) in sparrowhawk, mercury and cadmium in otter (2016), and PBT substances in harbour porpoise (2018).

3.1.1 Summary of available data

The data sources considered within the dashboard are outlined in Table 3.1.1.

Table 3.1.1 Data sources used for the H4 indicator[1]

Compartment	Media type	Species (common name)	Source organisation	Monitoring scheme
Terrestrial	Biota	Eurasian sparrowhawk	UKCEH	PBMS
Terrestrial	Biota	Red kite	UKCEH/Fera Science	PBMS/WIIS
Terrestrial	Biota	Red fox	Fera Science	WIIS
Freshwater	Water	–	Environment Agency	Statutory water quality monitoring surveillance programme/Watch List sites/CSF monitoring
Freshwater	Biota	Brown trout, chub and roach	Environment Agency	Statutory water quality monitoring biota – fish PBT substance and fish trend programmes
Freshwater	Biota	Eurasian otter	Cardiff University	Cardiff University Otter Project
Marine	Water	–	Environment Agency	Statutory water quality monitoring surveillance programme
Marine	Biota	Blue mussel	Environment Agency	OSPAR/Statutory water quality monitoring biota – mussels
Marine	Biota	Dab	Cefas	MSFD–OSPAR
Marine	Biota	Harbour porpoise	Cefas	CSIP and Cefas SLA with Defra

[1]UKCEH: UK Centre for Ecology and Hydrology; Cefas: Centre for Environment, Fisheries and Aquaculture Science; PBMS: Predatory Bird Monitoring Scheme; WIIS: Wildlife Incident Investigation Scheme; CSF: Catchment Sensitive Farming; OSPAR: Oslo and Paris Convention; MSFD: Marine Strategy Framework Directive (UK Marine Strategy); CSIP: UK Cetacean Strandings Investigation Programme; SLA: service level agreement.

The substances and time periods covered by the data are summarised in Table 3.1.2. The data are from the last two decades and run to the end of 2019, where possible. Data may be affected by changes in monitoring regimes over time. Any previous changes to sampling and analysis have been noted within the subsections of Section 4, as well as any influence that has had on the year selection for that dataset.

Table 3.1.2 Substances and time periods covered by the data in the H4 indicator[1]

Compartment	Species (common name) or media	Chemical group	Substance(s)	Time period
Terrestrial	Eurasian sparrowhawk	PBTs, heavy metals	Hg Pb Cd	2000, 2005, 2006, 2011–2013 2007–2014 2008–2014
Terrestrial	Red kite	Pesticides and biocides	SGARs	2015–2019
Terrestrial	Red fox	Pesticides and biocides	SGARs	2015, 2017–2019
Freshwater	Water	PBTs Heavy metals Pesticides and biocides	PFOS Pb, Cd, Ni, Cu, Zn Over 300 pesticides	2016–2019 2014–2019 2007–2019[2]
Freshwater	Brown trout, chub and roach	PBTs Heavy metals	Hg, PCBs PBDEs, PFOS Pb, Cd	2014–2019 2015–2019 2016–2019
Freshwater	Eurasian otter	PBTs Heavy metals	Hg PBDEs PFOS Pb Cd	2007–2009, 2014–2016 2003–2006 2007–2009 2007, 2008, 2014–2016 2006–2009, 2014–2016
Marine	Water	Heavy metals	Pb, Cd, Ni, Cu, Zn	2014–2019
Marine	Blue mussel	PBTs Heavy metals	Hg, PCBs PBDEs Pb, Cd, Ni, Cu, Zn	2011–2019 2015–2019 2011–2019

Compartment	Species (common name) or media	Chemical group	Substance(s)	Time period
Marine	Dab	PBTs	Hg, PBDEs, PCBs PFOS	2008–2019 2014–2019
		Heavy metals	Pb, Cd, Ni, Cu, Zn	2008–2019
Marine	Harbour porpoise	PBTs	PBDEs PCBs PFOS	2004–2008, 2010–2018 2004–2018 2001–2003, 2012–2018

[1]PBTs: persistent, bioaccumulative and toxic substances; Hg: mercury; Pb: lead; Cd: cadmium; SGARs: second-generation anticoagulant rodenticides; PFOS: perfluorooctanesulfonic acid; Ni: nickel; Cu: copper; Zn: zinc; PBDEs: polybrominated diphenyl ethers; PCBs: polychlorinated biphenyls. [2]Years from 2007–2018 used as a baseline.

The congeners covered under the two groups polybrominated diphenyl ethers (PBDEs) and PCBs may differ between datasets and the specific substances included in any assessment are noted within the relevant subsections of Section 4.

Another way that datasets of PBDEs and PCBs may differ slightly is in their treatment of values below the limit of detection (LoD), so-called non-detects. Concentrations of individual congeners are summed to give total values for PBDEs and PCBs. To do this, the congener concentrations that are below the LoD are given a negligible value. This approach is applied under different reporting regimes (for example, EC, 2009) although may be implemented in slightly different ways depending on the calculation tools. It does mean that environmental concentrations may be assumed to be lower than they are. Conversely, it avoids falsely elevated concentrations that would be seen by summing non-detect results treated as half than or equal to the face value of the LoD.

For substances other than PBDEs and PCBs, non-detect results are treated as half the face value of the LoD. For each dataset under Section 4, the handling of results below the LoD is explained in more detail.

In the trial of the dashboard approach for reporting, Shore et al. (2020) reported soil data from the Countryside Survey 2007 (UKCEH, 2007) for mercury, and also noted potential data for other metals from the same source. Due to the age of the survey and limitations in being able to determine trends over time and recent potential risk, we have not included this matrix in the 2021 reporting. We recognise the importance of this type of data for reflecting soil health as well as it being an entrance point for the potential movement of chemicals both up the terrestrial food chain and to other environmental compartments. We are looking at ways to include soil data as part of future reporting, using samples such as

those collected as part of the UKCEH National Capability [UK-SCAPE](link) (UK Status, Change and Projections of the Environment) programme.

3.1.2 Trend assessment

Statistically significant trends over time are indicated in the dashboard using arrows (Figure 3.1.1). A horizontal arrow indicates no change in concentrations over time, and upward and downward arrows denote increasing and decreasing trends in concentrations, respectively.

Our aim is to report exposure trends when there is a minimum of at least 5 full years of change within the data (6 independent sampling years). Where this requirement is not met but a threshold assessment has been made, the circle for that dataset in the dashboard is left blank in the centre.

We have reviewed historic data – see Table 3.1.2 for the time periods used – for this interim indicator to see if there are any trends within those datasets that meet the above criterion. The results of these trend analyses are used in the dashboard.

For a small number of datasets, if has been possible to assess trends for the full time period available – long-term trend – and based on 5 full years of change – short-term trend. Both results are noted in the report, but only the long-term trends are used for the dashboard. Long-term trends are potentially more meaningful, particularly for PBT substances for which policies are already in place to limit their input into the environment, but the resulting effects are slow because of the persistent nature of the chemicals. It may be that as management action takes place over the lifetime of the 25-YEP, such as on the remediation of water affected by abandoned metal mines, trends become visible in the short term for some substances.

In the case of top predators, measured concentrations of essential metals may be purely an indication of maintained physiological levels. For this reason, such species may not be good indicators of environmental change. However, as an interim, we have considered available data in relation to essential metals in top predators (for example, see Section 4.12) and have left blank entries in the dashboard when we are unable to report on these substances.

It should be noted that the trends assessment may mask local changes in environmental concentrations of chemicals. We have endeavoured to comment on any observations relating to this where possible within specific data sections in Section 4 and included any key findings in the dashboard summary (Section 3.2).

In relation to the blue mussel data, monitoring sites around the coast were first established to assess the trend in contaminants in biota over time in the UK Regional Seas and covered a range of geographical locations with varying background pressures. However, logistical challenges, including the disappearance of key *Mytilus* beds, has resulted in the alteration of the monitoring regime over time. We will need to give further consideration to the continued impact of such changes on the year-on-year balance of sites across

geographical and pressure gradients when assessing overall concentrations and trends across English waters.

3.1.3 Threshold assessment

Exposure concentrations for the most-recent year(s) of data – see Table 3.1.2 for the latest year(s) – are assessed against available thresholds to give an indication of risk of impairment or death to wildlife. The basis of the thresholds may differ for each substance/media entry and further details can be found in the different data sections of Section 4 and in the corresponding references.

In most cases, the assessment is based on monitored concentrations for one year; exceptions are those for concentrations of chemicals in dab where 2 years of data are used and for PFOS and metals in water which is based on 3 years. These exceptions reflect the monitoring schemes in place and allow a more-complete assessment across England and its coastline.

The results are graded according to how many sites or individual samples out of those examined are below or above the threshold. As the percentage above increases, the circles in the dashboard appear increasingly darker (Figure 3.1.1).

Where possible we have used statutory values for the threshold assessments, such as those under the Water Framework Directive (Standards and Classification) Directions (UK Government, 2015) which are used for water quality assessments, or values used as part of international assessments, for example OSPAR. Our focus is on wildlife and, therefore, the thresholds used in the indicator may not always be the same as those used under other reporting regimes where values for the protection of human health may be used instead. In the absence of a statutory or internationally used value for the protection of wildlife, we have used thresholds based on agreed guidance (EC, 2011a) and that have been subject to a review process.

Where threshold values are based on different tissues to those measured, we have highlighted any differences in the interpretation of the results.

Some of our data platforms are for species for which statutory threshold values for chemicals do not exist. In such cases, we have used values from primary literature or reviews of such information. It should be noted that their derivation will be slightly different to statutory values where additional factors may have been applied to address uncertainty. Therefore, it is possible that concentrations reported as below a threshold for some data sources may reflect differences in the derivation of such values; this has not been assessed as part of this report.

In the assessment for SGARs in red kite, autopsy information was used (Section 4.24.4).

In the case of PBDEs and PCBs, thresholds are available for these substances as a group or for individual congeners. If each congener is assessed against a corresponding threshold, the result indicating the greatest risk is used in the dashboard.

We have sought to apply thresholds in a way that is as consistent as possible across compartments and appropriate to the data being assessed. Our risk assessment is a general guide to determine where further investigation or action may be advised and is not a compliance assessment.

Future work is needed to consider negligible or background levels for the substances in different matrices in the indicator. This will help differentiate between those substances which report no change but still require management and those that are consistently at negligible levels. It will also provide a route to decide when to remove substances from the indicator.

3.2 Dashboard summary

We have been able to report the H4 interim indicator using data representing 8 different matrices (Figure 3.1.1).

The indicator represents data for England with the exception of some of the marine data for fish and cetaceans. The fish data include a Welsh site in the Bristol Channel, thus still following the England coastline, and the cetacean data cover the UK reflecting the greater movement of these species. This UK approach is consistent with that adopted for other marine indicators under the Outcome Indicator Framework.

For all groups of substances, data gaps need to be addressed to get a fuller picture across compartments and improve our ability to report trends. Data for the terrestrial compartment are lacking.

Data are available for PBDEs, PFOS and lead in otters, lead and cadmium in freshwater fish, and SGARs in red fox. They are given in Section 4, but they are not adequate enough to allow reporting in the dashboard.

For those substances and matrices with available thresholds, concentrations are observed above these values. This is not an unexpected result given that these substances have been selected for the indicator as ones warranting attention because of their presence in the environment.

3.2.1 Persistent, bioaccumulative and toxic substances

This group covers Hg, PBDEs, PCBs, and PFOS. Recent data are available for PBT substances in freshwater fish and marine biota with the exception of mercury in cetaceans. Data for the terrestrial compartment and in otters are mainly historical (see Table 3.1.2) and limited, meaning that we are unable to report changes over time for those with the exception of mercury in sparrowhawk and otter liver.

Concentrations of the different PBTs do not appear to show any statistically significant trends in most media that could be assessed, with the exception of some of the marine data for the higher trophic levels (Figure 3.1.1). For PBDEs in dab, a strong downward

trend was seen and this was the same observation in harbour porpoise. The same trend for PCBs was less pronounced and in dab only. Levels of PFOS in harbour porpoise are decreasing; however, only limited data are available for other media and more information is needed to establish a picture for this substance.

Based on our consideration of risk owing to PBTs, mercury shows the greatest exceedance of available thresholds for mussels and freshwater and marine fish (Figure 3.1.1). The result for mercury in dab is a tentative one as it is based on a threshold that could be considered over-precautionary for the tissue – muscle rather than whole fish – examined (see Section 4.8.4).

PCBs also are found at levels above their corresponding thresholds in the marine environment (Figure 3.1.1) and their slower decline suggests these substances may also be a higher priority. Because different data sources monitor different congeners, it may be useful to assess exposure to the commonly monitored congener PCB118 in future work. This congener is one of the most prevalent in the environment and has driven the risk assessment results in the dashboard for those cases where individual congeners have been assessed.

For PFOS, a small number of sites show levels above the thresholds for fish and water (Figure 3.1.1). For PBDEs, the situation is more promising, showing only low proportion of samples exceeding the threshold for dab (Figure 3.1.1), though the picture will be improved with more data for the terrestrial and freshwater compartments.

3.2.2 Heavy metals

This group covers the metals lead, cadmium, nickel, copper, and zinc. Data are available for metals in water, mussels and marine fish up to 2019 (Table 3.1.2); data are available for lead and cadmium in freshwater fish but are insufficient to report in the dashboard. Data are also available for sparrowhawks and otters, though only up to 2014 and 2016, respectively.

In most cases, trend assessment was possible and shows fairly level concentrations (Figure 3.1.1). This may be expected for nickel, copper and zinc in higher predators if they are exhibiting normal physiological levels (see Section 3.1.2). However, statistically significant downward trends were observed for nickel and zinc in sparrowhawks. In contrast an upward trend was observed for nickel in dab; this was driven by marine sites in the east and south.

Risk assessment showed levels above the threshold at between 1 and 24% of sites for all metals in freshwater and for nickel and zinc in estuarine and coastal waters (Figure 3.1.1). It should be noted that the nickel exceedance for marine was driven by one site in the south west. Out of the metals, zinc showed the highest rate of threshold exceedance in both freshwater and estuarine and coastal waters.

While the freshwater data for metals show no statistically significant change in concentrations from 2014 to 2019, these results can be split into two types: those for

waters affected by abandoned metal mines and those for sites in other locations. These data types are not distinguished between in the dashboard but are examined in Sections 4.13 and 4.14. Cadmium and copper concentrations show downward trends for the 'other' sites over the assessed time period. For waters affected by abandoned metal mines, their elevated levels of metals mean that they comprise a high proportion of those sites which exceed available thresholds, except for nickel where sites in other locations comprise the majority of those at risk.

3.2.3 Pesticides and biocides

This group covers pesticides and SGARs. It is not possible to assess trends currently for these chemicals.

The dataset for pesticides in freshwater is based on scan data which cover a broad range of substances, particularly compared with the number historically monitored using traditional quantitative methods. Sampling at the sites in the assessment has increased over time; data for years to 2018 have been grouped as a baseline. There are too few data to assess any trends over time for red kite or red fox.

Threshold exceedance is indicated for less than a quarter of sites or individuals considered for pesticides in freshwater and SGARs in red kite. For pesticides in freshwater, the threshold exceedance is driven by one site. It is planned to change this assessment in future so that it considers risk from chronic exposure.

4 Underlying data considered in the dashboard

The underlying data considered for each of the entries in the dashboard are described in the following subsections. Each of these data reports is presented in a similar format covering the following:

- The source of the data
- The structure of the dataset considered – for example: years considered, sampling regime, units of analysis, treatment of results below the LoD
- Presentation of the data and consideration of trends in chemical concentrations over time
- Proposed threshold for use and its comparison against the most recent data

The translation of the results into the metrics in the dashboard is briefly explained in the relevant sections on trends and threshold comparison.

The running order for the data report subsections mirrors the dashboard entries made for each contaminant, progressing through the different trophic levels in the terrestrial, then freshwater and finally marine compartments before moving on to the next contaminant down. However, substances are grouped in some cases for a data source, particularly for PBT substances and metals, to reduce repetition within the report.

For cases where 2019 data are not included, the information given here generally reflects that already provided in Shore et al. (2020). Some modifications to the previous assessment have been made for this report based on new decisions on minimum data requirements for trend assessment (Section 3.1.2) and how to apply available thresholds.

4.1 Persistent, bioaccumulative and toxic substances in sparrowhawk (*Accipiter nisus*) liver: mercury
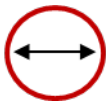
4.1.1 Data source

Data on total mercury in sparrowhawk livers is provided by the Predatory Bird Monitoring Scheme (PBMS) (UKCEH, 2020) from which the UKCEH has reported concentrations of mercury in liver in a series of reports (Shore et al., 2005; Walker et al., 2007, 2014, 2016).

Livers were collected from sparrowhawks found dead in each year. The birds had died from various causes (mainly collision or starvation) considered generally unrelated to their chemical exposure. Liver mercury concentrations have been reported for a number (but not all years) between 2000 and 2013. Birds collected by the PBMS were from Great Britain.

The data used for the time-trend analysis for the dashboard represent a sub-sample of these birds, specifically first year, non-starved females that were found dead throughout England. First year birds, so defined as individuals hatched in the current or previous year to that in which they were found dead, were used as they are likely to provide a more sensitive measure of annual change in exposure than adults. This is because adults may be exposed to, and bioaccumulate, mercury in the liver over multiple years.

Birds assessed at post-mortem to be in a starved condition were also excluded as starvation can mobilise mercury from other parts of the body into the liver and result in relatively elevated concentrations compared with those in nonstarved birds (Wienburg and Shore, 2004). Variation in the nutritional condition of birds between years may obscure the detection of trends in exposure over time, hence the decision to use first-year birds. Females were chosen over males because, although residues are typically higher in males than females, those in males also appear to be more variable (Walker et al., 2016) and therefore less sensitive for detecting annual changes in concentrations.

For the assessment of threshold exceedance, all birds analysed irrespective of age and whether they were in a starved state or not were included in the analysis. This is because the assessment focusses on looking at potential risk owing to levels of contamination so mercury concentrations in all birds are relevant.

4.1.2 Data structure

The data consist of measurements of mercury in the liver of a variable number individuals that died each year for the years 2000, 2005, 2006, 2011–2013. Data are reported as µg/g dry weight. The LoD for mercury was 0.09µg/g dry weight. All samples analysed had mercury concentrations above the LoD.

4.1.3 Exploration of change in chemical concentrations over time

The distribution of data by year for first-year, non-starved female birds is shown in Figure 4.1.1 and is summarised in Table 4.1.1.

Figure 4.1.1 Scatterplot of mercury (Hg) residues in the liver of first-year, non-starved female sparrowhawks from England. Data for individuals are shown. Horizontal red lines indicate median values by year (diagram courtesy of UKCEH)
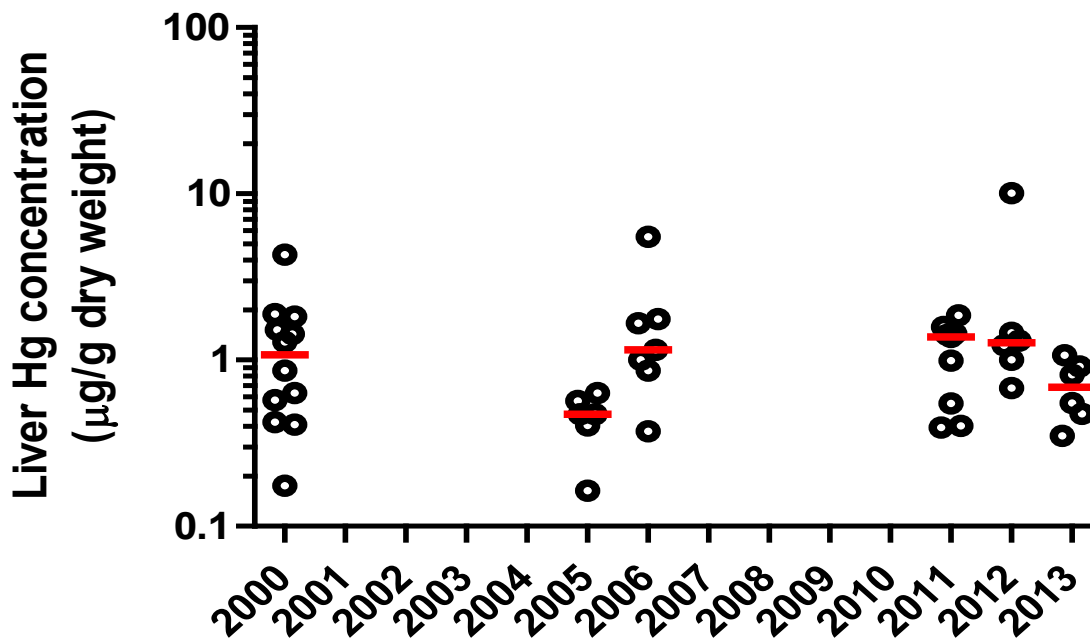

Table 4.1.1 Summary statistics for mercury concentrations in the liver of first-year, non-starved female sparrowhawks (µg/g dry weight)[1]

Year	n	Mean	SD	Median	Min	Max	Q1	Q3
2000	12	1.28	1.11	1.07	0.18	4.30	0.46	1.75
2005	6	0.45	0.16	0.47	0.16	0.63	0.34	0.58
2006	7	1.76	1.72	1.15	0.37	5.50	0.86	1.76
2011	9	1.11	0.55	1.38	0.39	1.86	0.47	1.51
2012	6	2.63	3.67	1.27	0.68	10.10	0.92	3.62
2013	6	0.70	0.28	0.68	0.35	1.07	0.44	0.95

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1 lower interquartile range value; Q3: upper interquartile range value.

Data post-2013 are not currently available and so trends over time could be assessed only for those years up to 2013 with available data.

Detailed analyses of trends over time of mercury concentrations in the livers of sparrowhawks are reported by Walker et al. (2016). They found no statistically significant upward or downward trends over a wider time period (1990–2013) for all sparrowhawks, taking age and sex into account as factors in the analysis. Re-analysis of our smaller dataset likewise found no statistically significant upward or downward trend over the period 2000–2013 (regression analysis: $F_{1,44}$ = 0.19, p >0.05). Therefore, the assignment of no change in concentrations (↔) is given in the dashboard.

4.1.4 Thresholds

There are no established threshold or EQS values for mercury in sparrowhawk livers, but Shore et al. (2011) proposed a minimum indicative mercury concentration in liver of 2µg/g wet weight, above which adverse effects on reproduction may occur in non-marine bird populations.

This value is based on the lowest species geometric mean for residues that have been associated with impaired reproduction including a range of effects, but predominantly decreased egg hatchability. The analysis is based on data for multiple species including ring-necked pheasants (*Phasianus colchicus*), mallard (*Anas platyrhynchos*), tree swallows (*Tachycineta bicolor*), and house wrens (*Troglodytes aedon*), with the lowest geometric mean observed in ring-necked pheasants. It is not indicative of a threshold for effects in individual birds. Therefore, for the indicator, the threshold is compared against the average (geometric mean or median) results for all birds sampled in a year rather than values for individuals.

Using a mean wet weight to dry weight conversion factor for sparrowhawks of 3.52[1], the indicative threshold concentration is equivalent to 7µg/g dry weight.

The summary data for all birds for which mercury residues in their liver were assessed against the threshold are given in Table 4.1.2.

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[1] R.F. Shore unpub. data; mean (± Standard Error) of 3.52± 0.02 based on measurements on 1454 livers.

Table 4.1.2 Summary statistics for mercury concentrations in the liver of all sparrowhawks (µg/g dry weight)[1]

Year	n	Mean	SD	Median	Min	Max	Q1	Q3
2000	63	4.62	4.66	2.42	0.18	19.32	1.19	6.30
2005	31	2.36	2.04	1.71	0.16	8.44	0.94	3.56
2006	30	2.58	2.34	1.62	0.16	9.11	0.89	3.79
2011	30	1.45	1.15	1.30	0.05	6.13	0.83	1.81
2012	26	1.82	2.30	1.18	0.05	10.10	0.80	1.70
2013	25	2.36	2.60	1.24	0.12	10.00	0.72	3.31

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1 lower interquartile range value; Q3: upper interquartile range value.

The median value for mercury in all sparrowhawks across all the years is 1.58µg/g dry weight and all annual medians are below the proposed threshold.

The entry in the dashboard is based on the results for the most-recent year available, 2013. This year had a median value approximately 6-fold lower than the proposed threshold mercury concentration in liver of 7µg/g dry weight. Therefore the entry reads 'All sites/individuals or population average below threshold'.

Given that the threshold is based on reproduction effects, it may be appropriate to confine or compare this assessment to one that considers females only in future. Based on individual results, only 6 females showed concentrations higher than the threshold across all years.

4.2 Persistent, bioaccumulative and toxic substances in freshwater: perfluorooctanesulfonic acid

4.2.1 Data source

Data on PFOS in water have been provided by the Environment Agency from their freshwater statutory monitoring network. Unlike the other PBT substances considered under this indicator, PFOS is water soluble and there is analytical capability to detect it in freshwater (Environment Agency, 2019).

4.2.2 Data structure

Data are available for the period 2016–2019 for PFOS in freshwater samples taken across England. The data vary both in terms of the number of measurements taken within a year per site and the number of sites monitored per year.

Some sites have been sampled in multiple years, although there were fewer sites and samples in 2019. The previous years' monitoring was greater to support the establishment of an evidence base for risk assessment and classification based on the analysis of PFOS in water and fish. Freshwater sites were limited following that, though still include those at which fish monitoring is also undertaken (see Section 4.3).

A data summary is available for each year based on the total number of measurements made in a year – that is all data pooled from all sites (see Table 4.2.1). Summaries are also available for each site based on samples taken over the most recent 3 years and for which there were more than 3 samples per year.

The units of measure were µg/l. The LoD for samples varied from 9 x 10^{-5} to 0.03µg/L. More than 99% of results were above the LoD. Results recorded as below the LoD were assigned a value equal to half the LoD.

4.2.3 Exploration of change in chemical concentrations over time

A summary of how PFOS concentrations in freshwater in England have varied over time is presented in Table 4.2.1 and Figure 4.2.1. The presentation of data in the figure differs to that for some of the other data sources in this report because the large number of samples for which data are available would otherwise result in a cluttered figure. For the purposes of clarity, the data are presented as annual median, interquartile range and 10–90th percentiles of individual sample concentrations taken.

Table 4.2.1 Summary statistics for concentrations of perfluorooctanesulfonic acid in freshwater (ng/L)[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2016	214	1701	6.04	9.56	3.43	0.180	172	1.71	6.58
2017	337	1909	6.23	15.5	2.92	0.120	428	1.27	6.44
2018	308	2305	8.39	66.3	3.17	0.0450	2500	1.39	6.33
2019	161	784	5.88	26.7	2.61	0.0450	711	1.12	6.29

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1 lower interquartile range value; Q3: upper interquartile range value.

Figure 4.2.1 Median, interquartile range and 10–90th percentiles of perfluorooctanesulfonic acid concentrations in freshwater (µg/L)
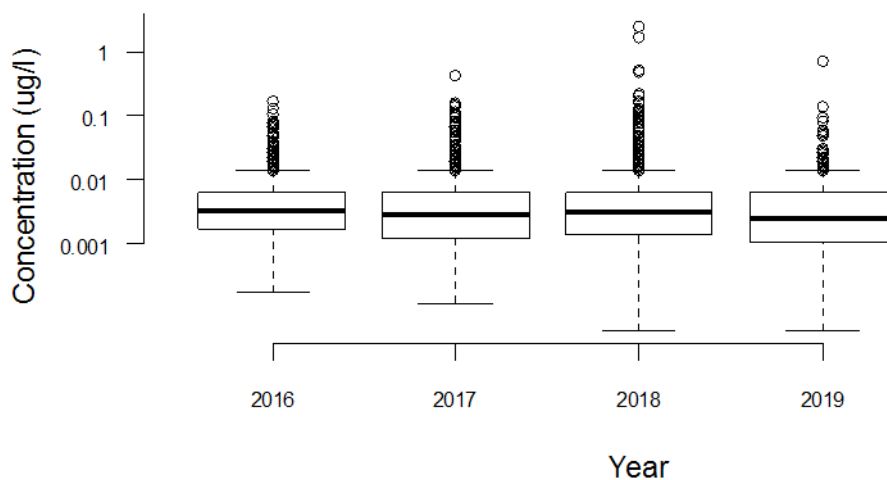
Because the data do not meet the minimum requirements for trend reporting, a formal trends assessment has not been performed. The entry on the dashboard relating to trends is blank.

4.2.4 Thresholds

To consider the risk to freshwater wildlife from PBT substances, secondary poisoning quality standards (QS$_{sec pois}$) have been used. These standards help protect wildlife from the effects of eating prey contaminated by PBT substances. A QS$_{sec pois}$ for PFOS of 33µg/kg wet weight (EC, 2011) has been derived through the EU EQS derivation process, which considers different protection goals. The QS$_{sec pois}$ for PFOS does not have statutory status as an EQS because it is not the most critical (lowest) QS. The EQS has a different
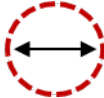
protection goal of human health; however the $QS_{sec\ pois}$ is the most appropriate to use here.

An equivalent empirical water concentration value has been derived from the $QS_{sec\ pois}$ by the Environment Agency for PFOS based on the relationship seen between monitored concentrations of PFOS in water and in fish (see Appendix A). This value is 0.019µg/L.

Typically, average site concentrations are used for comparison with the threshold for PFOS. These are based on available data for the most-recent 3 years. Thus, the starting year may vary by site as well as the number of years of data available. Each site requires more than 3 samples taken over that period to be included in the assessment; the number of samples per site varied between 4 and 37.

Twenty-one out of 463 sites (4.5%) had mean concentrations of PFOS in freshwater above the threshold of 0.019µg/L. The percentage result is used as the entry for the dashboard.

4.3 Persistent, bioaccumulative and toxic substances in freshwater fish: mercury, polybrominated diphenyl ethers, polychlorinated biphenyls and perfluorooctanesulfonic acid

Mercury

Polybrominated diphenyl ethers

Polychlorinated biphenyls

Perfluorooctanesulfonic acid

4.3.1 Data source

Data on mercury, PBDEs, PCBs, and PFOS in fish in England have been provided by the Environment Agency. Concentration data in whole fish (roach, chub and brown trout) have been collected by the Environment Agency as part of its biota monitoring, which began in anticipation of requirements under the Water Environment Regulations 2017 (UK Government, 2017).

4.3.2 Data structure

Relevant data are available for the period 2014–2019 for total mercury and PCBs and 2015–2019 for PBDEs and PFOS. The data consist of a variable number of measurements of the substances in terms of the number of freshwater fish sampled at a site and the number of sites monitored per year across England.

Individual sites are sampled once in a year. Typically, 5 fish are collected and analysed; however, the numbers in the past have varied from 2 to 10 fish.

Some sites have been sampled in multiple years and this varies for each contaminant. It should be noted that this data source is relatively new and a baseline dataset relating to designated trend sites is still being established. We have considered all site data as part of this assessment. This includes wider monitoring that was undertaken in 2018.

A data summary is available for each year based on the total number of measurements made in a year – that is all data pooled from all sites (see Tables 4.3.1, 4.3.2, 4.3.3, and

4.3.4). Summaries are also available for each site per year for those sites that had more than 1 sample per year.

For PBDEs, concentration data are the summed concentrations of six PBDE congeners, specifically PBDEs 28, 47, 99, 100, 153, and 154 (SUM 6PBDEs).

For PCBs, the Environment Agency collect data on congeners that are known for having the same mode of toxic action as harmful polychlorinated dibenzo-*p*-dioxins and dibenzofurans. These 12 dioxin-like congeners are PCBs 77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169, and 189. The PCB concentration data have been summed as their corresponding total TEQ (toxic equivalent) value for each sample for the purpose of assessing trends (Section 4.3.3). This total PCBs-TEQ value takes into account the relative toxicity of the different congeners so that their summed concentrations over time reflect those most likely to represent a risk to wildlife rather than looking at change in abundance alone.

All reported mercury concentrations were above the LoD. For PFOS, where concentration values have been reported below the LoD – generally set at 1μg/kg wet weight – they were assigned a value that was half the LoD. This related to approximately 4% of results.

For PBDE congeners, values below the LoD were assigned a negligible value of 0.00000001μg/kg wet weight for each congener. For PCBs, less than values were assigned as zero for calculating the total PCB-TEQ value.

4.3.3 Exploration of change in chemical concentrations over time

A summary of how mercury, SUM 6PBDEs, total PCBs-TEQ, and PFOS concentrations have varied over time is presented in Tables 4.3.1, 4.3.2, 4.3.3, and 4.3.4 and in Figure 4.3.1.

For PBDEs and PFOS, there are too few data to report any trend assessment currently. Analysis of the available data showed no statistically significant change (p >0.05) over time.

Only mercury and PCBs met the minimum data requirements for reporting a trend assessment. For the purposes of providing national trend information for the dashboard, all individual fish samples analysed were considered in the assessment of trends in concentrations over time.

The geometric mean was taken of all samples at all sites per year for mercury and PFOS to minimise any skews in the data and the undue influence of outliers. The tseries package in R was used to analyse the time series and identify any potential trends, the significance of which was assessed using the Cox Stuart trend test. Statistically significant trends were those for which the p-value was <0.05. Assessments based on medians and geomeans of sites and were also performed but there was very little difference in the results.

Table 4.3.1 Summary statistics for concentrations of mercury in whole freshwater fish (µg/kg wet weight)[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2014	26	143	32.6	25.9	24.5	6.18	156	17.8	35.5
2015	27	127	37.1	23.6	30.1	6.39	133	21.1	46.5
2016	23	109	38.7	23.1	32.1	9.42	134	22.1	49.6
2017	21	99	33.2	20.6	25.9	7.78	97.4	19.1	39.4
2018	43	193	48.5	46.9	33.4	8.31	300	21.1	54.1
2019	29	137	39.3	39.6	28.1	10.7	326	19.9	46.4

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1 lower interquartile range value; Q3: upper interquartile range value.

Table 4.3.2 Summary statistics for concentrations of SUM 6PBDEs in whole freshwater fish (µg/kg wet weight)[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2015	27	127	7.34	5.56	5.97	0.155	26.1	2.89	9.65
2016	14	65	4.91	3.37	4.51	0.263	14.5	2.22	6.61
2017	13	62	4.71	4.49	3.25	0.287	24.8	1.83	5.90
2018	12	57	5.24	6.19	3.36	0.245	41.0	1.71	7.15
2019	24	116	3.18	2.45	2.90	0.236	12.0	1.41	4.19

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1 lower interquartile range value; Q3: upper interquartile range value.

Table 4.3.3 Summary statistics for concentrations of total PCBs-TEQ in whole freshwater fish (ng/kg wet weight)[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2014	5	18	0.622	0.620	0.412	0.0163	2.02	0.0341	0.958
2015	12	54	0.707	0.428	0.729	4.58×10^{-3}	1.80	0.435	0.993
2016	21	99	0.826	1.08	0.497	8.49×10^{-3}	8.12	0.210	1.07
2017	21	103	0.541	0.734	0.286	3.70×10^{-4}	6.00	0.135	0.732
2018	38	170	0.646	0.953	0.381	1.61×10^{-3}	8.16	0.222	0.715
2019	27	128	1.30	1.23	1.00	2.60×10^{-4}	5.31	0.365	1.87

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1 lower interquartile range value; Q3: upper interquartile range value.

Table 4.3.4 Summary statistics for concentrations of perfluorooctanesulfonic acid in whole freshwater fish (µg/kg wet weight)[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2015	7	33	12.0	7.88	8.92	2.71	25.9	4.11	20.0
2016	22	104	14.0	11.2	10.0	0.500	47.5	4.97	20.1
2017	20	95	12.1	10.5	9.93	0.500	43.8	3.10	15.1
2018	43	195	11.4	12.3	6.17	0.500	57.4	3.51	13.9
2019	28	132	14.4	15.1	9.11	0.500	69.0	4.33	20.5

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1 lower interquartile range value; Q3: upper interquartile range value.

Figure 4.3.1 Median, interquartile range and 10–90th percentiles of mercury (Hg), SUM 6PBDEs, total PCBs-TEQ, and perfluorooctanesulfonic acid (PFOS) concentrations (µg/kg wet weight, except for total PCBs-TEQ in ng/kg wet weight) in whole freshwater fish
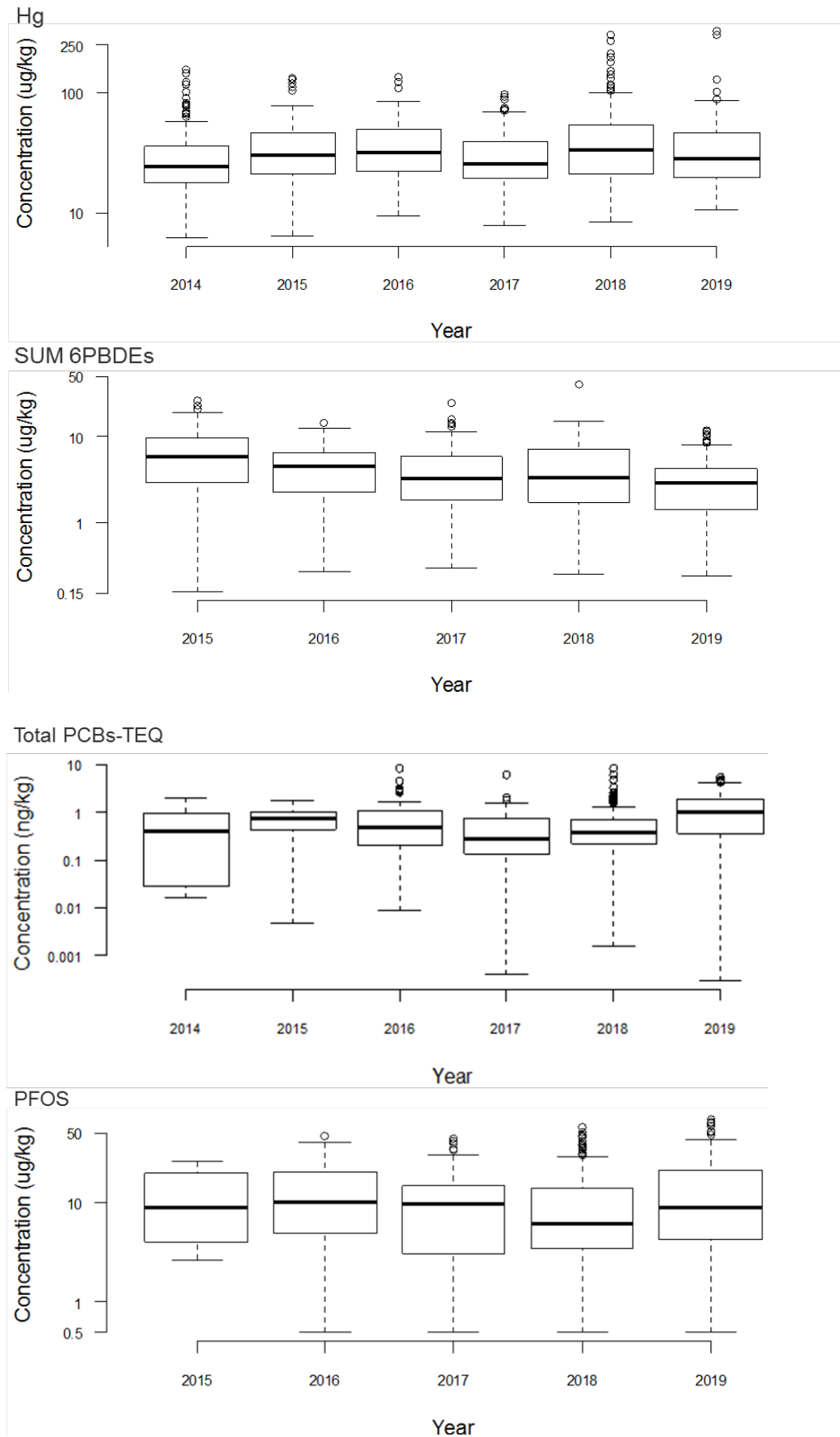
Assessment of the data shows that there are no statistically significant trends based on the geometric means for mercury and total PCBs-TEQ (Table 4.3.5). The dashboard trend information is based on the overall national assessment; therefore, the corresponding entry is 'No observed change in concentrations' for mercury and PCBs. For PBDEs and PFOS, the trend information is left blank to indicate that the minimum requirements for reporting trends are not met.

Table 4.3.5 Summary of p-values from the assessment of trends over time based on the geometric means of mercury and SUM PCBs-TEQ concentrations in freshwater fish

Substance	p-value Any trend	p-value Downward trend	p-value Upward trend	Decision
Mercury	1.0	0.88	0.50	no change
SUM PCBs-TEQ	1.0	0.50	0.88	no change

To improve our picture of what is happening in relation to PCBs across different environmental compartments and trophic levels, we may want future trend assessments to consider changes over time in the levels of PCB118. There are more datasets available for this congener as it tends to be common across different reporting regimes. This is because it is a substance that is likely to be present in the environment, owing to its relatively high use in technical mixtures in the past (Agency for Toxic Substance and Disease Registry (ATSDR, 2000) and persistent and bioaccumulative behaviour. However, there is value in continuing to report changes for SUM PCBs-TEQ alongside this for the reasons outlined in Section 4.3.2 and to give a better idea of the magnitude of exposure to PCBs in general.

4.3.4 Thresholds

To consider the risk to freshwater wildlife from PBT substances, secondary poisoning quality standards ($QS_{sec\ pois}$) have been used. These standards help protect wildlife from the effects of eating prey contaminated by PBT substances.

The EQS specified in the Water Framework Directive (Standards and Classification) Directions 2015 (UK Government, 2015) for mercury is a $QS_{sec\ pois}$ and is 20μg/kg wet weight in fish.

For PBDEs and PFOS, $QS_{sec\ pois}$ values have been derived through the EU EQS derivation process, which considers different protection goals. The $QS_{sec\ pois}$ values for PBDEs and PFOS do not have statutory status as EQSs because they are not the most-critical (lowest) QSs. The EQSs have a different protection goal of human health; however the $QS_{sec\ pois}$ values are the most appropriate to use here. The derived $QS_{sec\ pois}$ for PBDEs is 44μg/kg wet weight (EC, 2011c) and that for PFOS is 33μg/kg wet weight (EC, 2011b).

There are no standards available for solely dioxin-like PCBs.

Average concentrations for each substance or group of substances from sites assessed in 2019 were compared against the above values. Sites required more than 1 sample to be included in the assessment. The results are summarised in Table 4.3.6 and information on the percentage of sites above the corresponding thresholds was used for the dashboard.

Results for mercury suggested the greatest risk to wildlife.

For PCBs, the entry in the dashboard reflects that there are no values available for comparison.

Table 4.3.6 Summary of threshold comparison information for freshwater fish for 2019

Substance	Number of sites	Number of sites above threshold	Percentage of sites above threshold (%)
Mercury	29	23	79
PBDEs	24	0	0
PFOS	28	4	14

It should be noted that because of potential differences in protection goals and methods used for national assessments, the results reflected in the dashboard may differ slightly to those used for water quality classification and reporting purposes.

4.4 Persistent, bioaccumulative and toxic substances in Eurasian otter (*Lutra lutra*): mercury
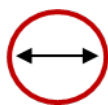
4.4.1 Data source

Data on total mercury in otter livers have been provided by the Cardiff University Otter Project (CUOP) (Cardiff University, 2020). Livers have been collected from individuals found dead each year. Most animals died as a result of traffic collisions but some individuals died from other causes.

4.4.2 Data structure

The data consist of measurements of mercury concentrations in liver from carcasses found each year from 2007 to 2009 and from 2014 to 2016 from a stratified random sub-sample[2] of all animals collected from England and Wales. Only data for samples from England are considered in this report.

Samples were analysed in three tranches that is 2007–2008, 2009 and 2014–2016, with the earlier two tranches having a higher LoD of 0.2µg/g dry weight than the later tranche. Therefore, a common limit of detection of 0.2µg/g dry weight was applied to the whole dataset, resulting in three samples being below the common LoD. These samples were assigned a value of 0.1µg/g dry weight (half the LoD) for statistical analysis.

The data for mercury concentrations in liver were not normally distributed for some years and across the whole dataset, either as measured concentrations or log10-transformed data; therefore, non-parametric descriptive statistics are reported here.

4.4.3 Exploration of change in chemical concentrations over time

The distribution of data over time is shown in Figure 4.4.1 and is summarised in Table 4.4.1.

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[2] Overall sample set stratified so it was representative of age class, sex and year. Livers selected from animals at random from within those categories.

Figure 4.4.1 Scatterplot of mercury residues in the liver of Eurasian otters from England. Data shown are for individuals. Horizontal lines within plots indicate annual median values (diagram courtesy of UKCEH)
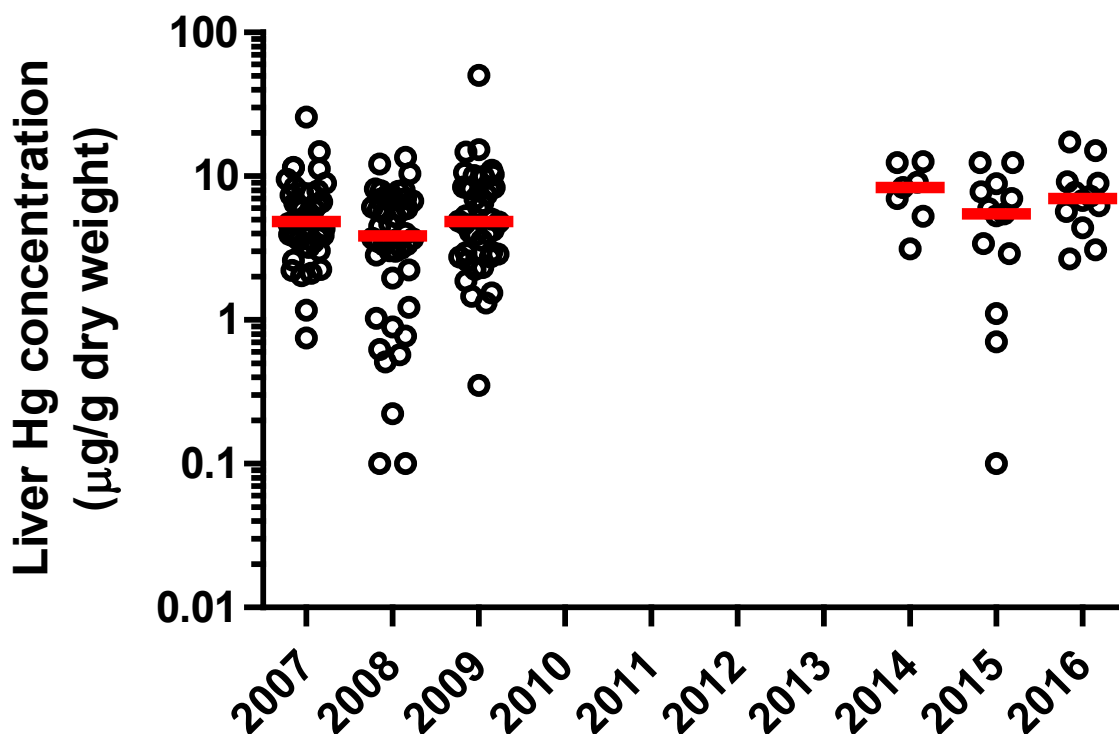
Table 4.4.1 Summary statistics for mercury concentrations (µg/g dry weight) in the liver of Eurasian otters from England[1]

Year	n	Mean	SD	Median	Min	Max	Q1	Q3
2007	40	6.062	4.504	4.830	0.749	25.90	3.325	7.655
2008	42	4.541	3.296	3.855	0.100	13.50	1.770	6.930
2009	40	6.825	7.928	4.850	0.352	50.30	2.780	8.313
2014	7	8.263	3.500	8.350	3.130	12.60	5.260	12.40
2015	13	5.647	4.047	5.490	0.100	12.40	2.005	8.325
2016	12	7.863	4.424	7.020	2.670	17.30	4.700	9.130

[1]n: number of samples analysed; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

The residual values examined using exploratory regression analysis of raw or log10-transformed data were not normally distributed and both visual inspection of the data and analysis using Spearman's rank correlation indicated that the mercury concentration in

liver and year were not correlated. Analysis of differences between years (where sample numbers exceeded 5 in a year) in mercury concentrations, using a Kruskal–Wallis test, indicated statistically significant differences between all years (KW = 10.82, p = 0.029) but Dunn's Multiple Comparison Test found no statistically significant difference between all possible pairs of years.

The apparent contradiction of results from these two tests may suggest that the result from Kruskal–Wallis test is spurious and so there is no clear evidence of a trend in concentrations over time from the data.

Because there was not a statistically significant trend in concentrations over time, the entry made on the dashboard is 'No observed change in concentrations (↔)' for mercury in otter liver.

4.4.4 Thresholds

There is no statutory threshold established for mercury concentrations in otter liver. However, Shore et al. (2011) suggest average liver concentrations in sampled populations that are greater than 25mg/kg wet weight may be indicative of some lethality and impaired reproduction in that population. This equates to 87.5mg/kg (or µg/g) dry weight when a wet weight to dry weight conversion factor of 3.5 is applied (Talmage and Walton, 1991). This value is an average for a sampled population and should be compared against an average (geometric mean or median value) rather than values for individuals.

The median value for mercury in the liver of otters that died in 2016 was 7.02µg/g dry weight, some 12-fold lower than the minimum indicative mercury concentration in liver of 87.5µg/g dry weight. The entry on the dashboard is therefore 'All sites/individuals or population average below threshold'.

4.5 Persistent, bioaccumulative and toxic substances in Eurasian otter (*Lutra lutra*): polybrominated diphenyl ethers

No dashboard entry – insufficient or no comparable data

4.5.1 Data source

Data on PBDEs in otter livers have been provided by the CUOP (Cardiff University, 2020). Livers have been collected from individuals found dead each year. Most animals died as a result of traffic collisions but some individuals may have died from other causes.

4.5.2 Data structure

The data consist of measurements of PBDE concentrations in liver from carcasses found each year between 2003 and 2006 from a stratified random sub-sample[3] of all animals collected from England and Wales. Only data for samples from England are considered in this report.

Values provided were for total PBDE concentrations in units of ng/g lipid weight based on quantification of 27 PBDE congeners, namely numbers 17, 28, 32, 35, 37, 47, 66, 71, 75, 77, 85, 99, 100, 119, 128, 138, 153, 154, 166, 183, 190, 196, 197, 206, 207, 208, and 209. Where individual congeners were not detected in samples, they were assigned a value of 0ng/g lipid weight for the purposes of summing congener residues to calculate total PBDE concentrations. All samples had a detectable concentration of total PBDEs.

Only summary data (means/medians) were available for assessment.

4.5.3 Exploration of change in chemical concentrations over time

The data available for assessment are summarised in Table 4.5.1.

An analysis of trend in concentrations over time cannot be performed as there are too few years of data. The most-recent year of data is 2006 and may not reflect the current levels of PBDEs in otter liver. The entry in the dashboard is blank and this, in part, reflects the lack of sufficient and current data.

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[3] Overall sample set stratified so it was representative of age class, sex and year. Livers selected from animals at random from within those categories

Table 4.5.1 Summary statistics for total polybrominated diphenyl ether concentrations in liver (ng/g lipid weight)[1]

Year	n	Mean	SD	Median	Min	Max
2003	7	7084	12904	987	307.5	35880
2004	28	1781	1454	1145	162.2	5814
2005	36	3155	3863	1997	12.2	20955
2006	19	2206	2470	1105	147.0	8324

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value.

4.5.4 Thresholds

A threshold has not been established for total PBDE concentrations in otter livers. Therefore, no threshold value is proposed for this metric. The blank entry in the dashboard, in part, reflects that there is no value available for comparison.

4.6 Persistent, bioaccumulative and toxic substances in Eurasian otter (*Lutra lutra*): perfluorooctanesulfonic acid

No dashboard entry – insufficient or no comparable data

4.6.1 Data source

Data on PFOS in otter livers have been provided by the CUOP ([Cardiff University, 2020](#)). Livers were collected from individuals found dead each year. Most animals died as a result of traffic collisions but some individuals died from other causes.

4.6.2 Data structure

The data consist of measurements of PFOS concentrations in liver from carcasses found each year between 2007 and 2009. These are data from a stratified random sub-sample[4] of all animals collected from England and Wales. Only data for samples from England are considered in this report.

PFOS was detected in all samples analysed and concentration values are given in units of μg/kg wet weight.

4.6.3 Exploration of change in chemical concentrations over time

A summary of the annual variation in PFOS concentrations is presented in Table 4.6.1.

An analysis of trend in concentrations over time cannot be performed as there are too few years of data. The most-recent year of data is 2009 and may not reflect the current levels of PFOS in otter liver. The entry on the dashboard is blank and this, in part, reflects the lack of sufficient and current data.

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[4] Overall sample set stratified so it was representative of age class, sex and year. Livers selected from animals at random from within those categories

Table 4.6.1 Summary statistics for perfluorooctanesulfonic acid concentrations in otter liver (μg/kg wet weight)[1]

Year	n	Mean	SD	Median	Min	Max
2007	13	1981	1813	1130	78.8	5800
2008	15	1839	1568	1670	403.0	6800
2009	14	776	469	745.0	93.8	1840

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value.

4.6.4 Thresholds

There are no established PFOS threshold concentrations in otter livers, and we are unaware of suitable generic threshold values for other mammalian species that could be applied to otters. Therefore, no threshold value is proposed for this metric. The blank entry in the dashboard, in part, reflects that there is no value available for comparison.

4.7 Persistent, bioaccumulative and toxic substances in blue mussels (*Mytilus edulis*): mercury, polybrominated diphenyl ethers, polychlorinated biphenyls

Mercury

Polybrominated diphenyl ethers

Polychlorinated biphenyls

4.7.1 Data source

Data on mercury, PBDEs and PCBs in blue mussels (*Mytilus edulis*) in England have been provided by the Environment Agency. Mussel data were originally collected as part of the UK-wide OSPAR Coordinated Environmental Monitoring Programme (CEMP), and analysis was later expanded in anticipation of monitoring requirements under the Water Environment Regulations 2017 (UK Government, 2017).

Data used in this assessment are also submitted, as part of the wider UK dataset, to the DOME (marine environment) data portal for the International Council for the Exploration of the Sea (ICES) (ICES, 2021a).

4.7.2 Data structure

Data on total mercury and PCB concentrations in *Mytilus* flesh are available for the period 2000–2019, but only data from 2011 onwards are included in this assessment to eliminate the impacts of historical changes on the monitoring programme. For PBDEs, relevant data are available from 2015 to 2019.

For PBDEs, data are available for six individual PBDE congeners – 28, 47, 99, 100, 153, and 154 – and the summed concentration of these congeners (SUM 6PBDEs). For PCBs, data are available for the ICES-7 PCBs as individual congeners and as summed concentrations (SUM ICES-7). The ICES-7 PCBs are seven congeners commonly found in the environment and designated by the ICES as an indicator of the degree of PCB contamination. These congeners are PCBs 28, 52, 101, 118, 138, 153, and 180 and are monitored under the OSPAR CEMP.

Wet weight concentrations were provided for all substances. Lipid weight concentrations were also given for the individual ICES-7 PCBs to allow threshold comparisons (see Section 4.7.4). Treatment of data below the LoD is explained in Section 4.7.3.

The monitoring methodology is described in the CEMP programme manual, the Green Book (British Oceanographic Data Centre (BODC, 2020). Where feasible sites are monitored annually, with a target of three samples – consisting of pooled individuals – collected at each site on each sampling occasion. Samples are collected in the winter/early spring to avoid any seasonal influence from spawning.

4.7.3 Exploration of change in chemical concentrations over time

The data available for mercury in all samples at all sites is summarised in Table 4.7.1 and in Figure 4.7.1. All samples analysed had concentrations above the LoD (0.5µg/kg wet weight).

Table 4.7.1 Summary of data for mercury in *Mytilus edulis* (µg/kg wet weight) from samples taken at monitored sites in England since 2011[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2011	17	47	35.8	21.5	30.0	8.53	85.1	19.3	43.8
2012	17	33	34.8	16.1	32.6	10.3	76.1	23.8	39.8
2013	17	51	22.5	10.0	23.1	4.92	44.0	13.8	29.7
2014	20	61	37.4	21.7	31.7	5.00	122	24.5	45.8
2015	19	57	28.6	17.9	21.5	10.6	87.2	17.1	36.8
2016	16	48	33.9	14.0	33.9	12.9	60.2	21.7	45.4
2017	15	45	29.5	12.0	31.0	10.3	49.0	19.1	39.5
2018	13	37	32.3	12.7	34.0	12.0	51.9	20.9	43.3
2019	16	46	38.4	19.2	39.0	13.3	83.6	24.9	49.7

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; median: min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Figure 4.7.1 Median, interquartile range and 10–90th percentiles of concentrations of mercury in *Mytilus edulis* (µg/kg wet weight) since 2011
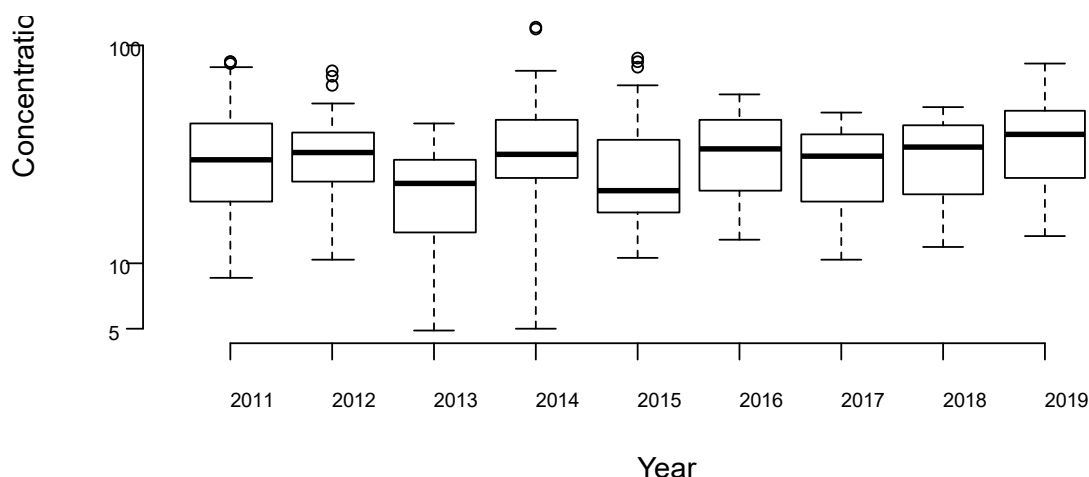
Temporal trend was assessed using the data for all individual samples analysed for mercury. The geometric mean was taken of all samples at all sites per year to minimise any skews in the data and the undue influence of outliers. The tseries package in R was used to analyse the time series and identify any potential trends, the significance of which was assessed using the Cox Stuart trend test. Statistically significant trends were those for which the p-value was <0.05. Assessments based on medians and geomeans of sites and were also performed but there was very little difference in the results.

There was no overall temporal trend found for mercury concentrations in *Mytilus* flesh (Table 4.7.2). Therefore, the corresponding entry in the dashboard is 'No observed change in concentrations'.

Table 4.7.2 Summary of p-values from the temporal trend assessment of the geometric means of mercury concentrations in mussels

p-value Any trend	p-value Downward trend	p-value Upward trend	Decision
0.63	0.94	0.31	no change

For PBDEs, summary data are provided for the summed concentrations of the six PBDE congeners analysed, SUM 6PBDEs (Table 4.7.3 and Figure 4.7.2). To generate the SUM 6PBDEs concentration, values below the LoD were assigned a negligible value of 0.00000001µg/kg wet weight for each congener. Where the SUM 6PBDEs was 0.00000006µg/kg wet weight, this value is also amended to 0.00000001µg/kg wet weight to ensure it remained a negligible value.

The LoDs and percentages of samples with values below the LoD varied between congeners (PBDE28: LoD 0.006µg/kg, 44%; PBDE47: LoD 0.02µg/kg, 3%; PBDE99: LoD

0.02μg/kg, 12%; PBDE100: LoD 0.008μg/kg, 4%; PBDE153: LoD 0.02μg/kg, 85%; PBDE154: LoD 0.01μg/kg, 29%).

Table 4.7.3 Summary of data for SUM 6PBDEs in *Mytilus edulis* (μg/kg wet weight) from samples taken at monitored sites in England since 2015[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2015	19	57	0.42	0.38	0.24	1.0×10^{-8}	1.5	0.13	0.64
2016	16	48	0.38	0.46	0.36	1.0×10^{-8}	3.2	0.14	0.50
2017	15	45	0.53	0.43	0.43	0.087	1.6	0.20	0.67
2018	13	39	0.34	0.26	0.29	0.033	0.91	0.11	0.51
2019	16	46	0.39	0.41	0.36	1.0×10^{-8}	2.2	0.13	0.53

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; median; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Figure 4.7.2 Median, interquartile range and 10–90th percentiles of SUM 6PBDEs concentrations in *Mytilus edulis* (μg/kg wet weight) since 2015
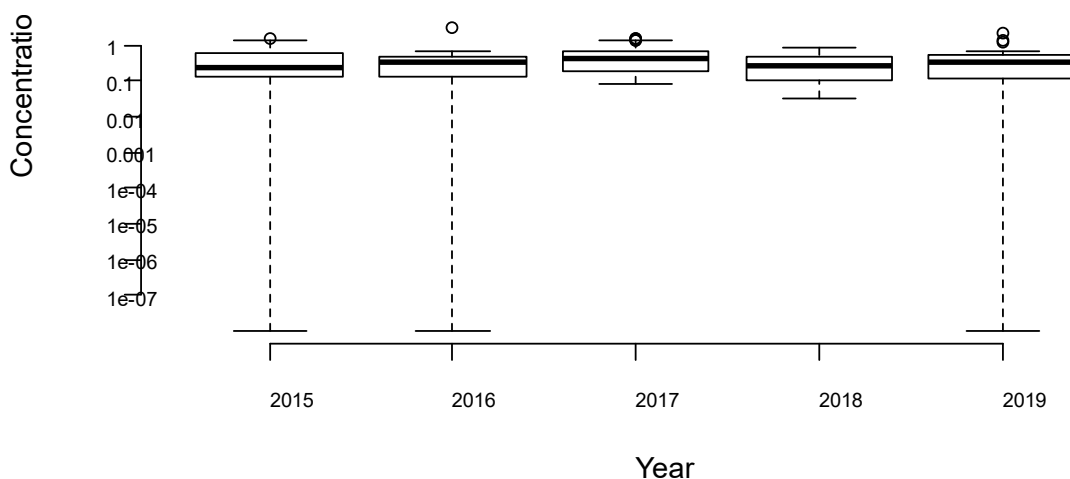
Simple visual inspection of the data shows a fairly level pattern of the concentration values throughout the years examined. Concentrations within the lower and upper interquartile range are within the same order of magnitude indicating that detected levels have remained fairly consistent.

As the PBDE data do not meet the minimum requirements for trend reporting, a formal trends assessment has not been performed. The entry on the dashboard relating to trends is blank.

Summary data available for SUM ICES-7 PCBs for each year are shown in Table 4.7.4 and Figure 4.7.3.

Any individual PCB result recorded below the LoD was set to the equivalent of near zero (0.00000001µg/kg wet weight) before being summed together to make the total. Where a sample recorded all 7 PCBs at below the LoD, the SUM ICES-7 PCBs has also been amended to 0.00000001µg/kg wet weight. Percentages of samples with values below the LoD varied between congeners (PCB28: 80%, PCB52: 70%, PCB101: 56%, PCB118: 58%, PCB138: 43%, PCB153: 23%, PCB180: 84%).

Table 4.7.4 Summary of data for SUM ICES-7 PCB in *Mytilus edulis* (µg/kg wet weight) from samples taken at monitored sites in England since 2011[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2011	17	47	1.12	1.76	1.00×10^{-8}	1.00×10^{-8}	6.72	1.00×10^{-8}	1.36
2012	14	30	1.11	1.05	1.22	1.00×10^{-8}	3.84	1.00×10^{-8}	1.75
2013	17	51	1.00	1.52	1.00×10^{-8}	1.00×10^{-8}	6.20	1.00×10^{-8}	1.60
2014	20	61	1.20	1.76	1.00	1.00×10^{-8}	8.69	1.00×10^{-8}	1.51
2015	19	57	3.90	4.36	2.32	0.13	18.6	0.75	6.25
2016	16	48	2.48	2.58	1.18	0.10	10.6	0.67	3.48
2017	15	45	3.26	2.74	3.28	0.12	8.92	0.75	4.51
2018	13	39	2.58	2.80	1.39	0.52	9.82	0.70	2.30
2019	16	46	2.20	2.20	1.30	1.00×10^{-8}	8.03	0.60	4.23

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; median; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Figure 4.7.3 Median, interquartile range and 10–90th percentiles of SUM ICES-7 PCBs in *Mytilus edulis* **(µg/kg wet weight) since 2011**
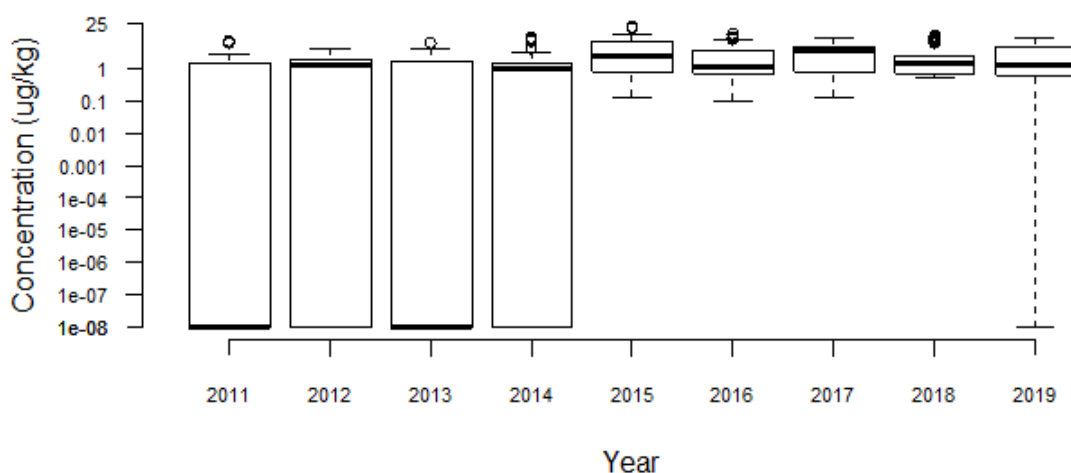
The analytical method was improved during 2014 so that the LoD for individual congeners changed by an order of magnitude from 1 to 0.1 µg/kg wet weight. From 2015 onwards, most individual congeners were reported as positive or below the improved LoD, although some reported less than results (5%) were censored at higher values (0.15–2 µg/kg wet weight).

Owing to significant change in the LoD over time, a formal trend assessment has not been performed for PCBs. The entry on the dashboard relating to trends is blank.

Consideration will be given to how best to evaluate the above data for PCBs for future indicator reporting. It may be preferable to focus on data from 2015 onwards owing to the improvement in analytical capability for those years. It is not clear whether the slightly higher results for these years are also owing to a different analytical technique or are because of an environmental change. Focus on PCB118 might also be appropriate to study the exposure pattern for a single congener known to be one of the most prevalent in the environment and to allow comparison across different datasets, as suggested in Section 4.3.3.

4.7.4 Thresholds

There is no EAC for mercury under OSPAR which covers the protection of wildlife. The EQS for mercury specified in the Water Framework Directive (Standards and Classification) Directions 2015 (UK Government, 2015) is derived to protect top predators from secondary poisoning ($QS_{sec pois}$). However it is based on fish which represent a different trophic level than *Mytilus* that is trophic level 4 rather than 2. To consider a level of trophic adjustment, it is proposed that an interim threshold value of 1.6µg/kg wet weight (OSPAR Commission, 2016) is used. However, the high uncertainty of converting an EQS in fish into an equivalent value in molluscs has to be recognised, as discussed in the OSPAR Commission report (2016).

There are no derived EACs for PBDE congeners under OSPAR, but possible EAC equivalents have been considered by the OSPAR Working Group on Monitoring and on

Trends and Effects of Substances in the Marine Environment (MIME). This resulted in the recommended use of Canadian Federal Environmental Quality Guidelines (FEQGs) (Environment and Climate Change Canada, 2013) for biota as EAC equivalents (ICES, 2021b). These are threshold values for individual PBDEs (Table 4.7.5), which correspond to the 6 congeners monitored in mussels. The $QS_{sec\ pois}$ derived through the EU EQS derivation process for SUM 6PBDEs (EC, 2011c) was considered by the OSPAR MIME, but rejected in favour of the individual thresholds to allow more-stringent assessment of the more-toxic congeners.

The EAC thresholds used in OSPAR assessments for the individual ICES-7 PCB congeners are used in this assessment. These are based on lipid weight (Table 4.7.5).

When performing the threshold assessment for each substance or group of substances, the most-recent site means, those for 2019, were assessed against the threshold. Only sites for which there was more than 1 sample were included; the assessment therefore comprised 15 of the available 16 sampling sites.

Table 4.7.5 Suggested values that can be used as dashboard thresholds for polybrominated diphenyl ethers and polychlorinated biphenyls

PBDE congener number	Canadian FEQG based values (µg/kg wet weight)	ICES-7 PCB congener number	OSPAR EAC (µg/kg lipid weight)
28	120	**28**	67
47	44	**52**	108
99	1	**101**	121
100	1	**118**	25
153	4	**138**	317
154	4	**153**	1585
–	–	**180**	469

For mercury, all site mean concentrations (100%), and indeed all individual samples, for 2019 exceeded the threshold. The entry on the dashboard reflects this result.

For the PBDEs, all site mean concentrations for 2019 for all individual congeners were below the threshold. The entry on the dashboard reflects this result.

The 2019 site mean concentrations for the individual PCBs showed no site exceedances for PCBs 28, 138, 153, or 180. The PCB52 EAC was exceeded by mean concentrations at a single site in the Mersey. The mean concentrations at two Mersey sites also exceeded the EAC for PCB101. At nine of the 15 sites monitored around the coast, the mean concentrations of PCB118 exceeded the EAC. Information on the percentage of sites (60%) above the PCB118 threshold was used for the dashboard.

4.8 Persistent, bioaccumulative and toxic substances in marine fish: mercury

4.8.1 Data source

Data on mercury concentrations are available for dab (*Limanda limanda*) muscle tissue. These data are collected as part of the Marine Strategy Framework Directive (UK Marine Strategy)–OSPAR (MSFD–OSPAR) monitoring for assessing good environmental status. The data are collected and held by the Centre for Environment, Fisheries and Aquaculture Science (Cefas).

Data used in this assessment are also submitted, as part of the wider UK dataset, to the DOME (marine environment) data portal for the ICES (ICES, 2021a).

Sites are selected on the basis that they reliably support dab populations that can be sampled for analysis. There are a minimum of three sites within each OSPAR hydro-geographical sub-region (OSPAR Commission, 2020) and there are no direct impacts from local sources.

Data have been provided for 2008–2019 covering 25 stations, although monitoring at one of these was only conducted in 2014. Between 2008 and 2010, sampling around the country was done annually and covered 16–23 stations each year. From 2011 onwards, fish were collected at east and west coast stations on alternate years. For these years, there are data for 14–15 east coast stations (odd years) and 8–9 west coast ones (even years).

All data relate to designated English waters, with the exception of those from a Welsh station in the Bristol Channel as this is a shared water body in which fish are likely to move freely across territorial waters.

Typically, 5 or fewer pools of fish were sampled around each station. Each pool comprised 5 fish. Stations at which there was only one pool collected have been excluded from the analysis.

4.8.2 Data structure

Data are for total mercury[5] in muscle. All concentration data are reported in mg/kg wet

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[5] All mercury species transformed to elemental mercury and the concentration is then determined.

weight. The data summaries that were provided consisted of results from individual pool samples taken around the stations. These individual samples were used in the trend and threshold assessments rather than station means for the purposes of the dashboard indicator. This is because the pooled samples are already representative of a mean of 5 fish and this approach allows assessment of trends across the stations and nationally.

The LoD varied across the dataset of 830 samples. Only 6 samples were reported below LoDs of 0.01 and 0.02mg/kg wet weight and these were assigned a value of 0.005 and 0.01mg/kg wet weight, respectively (half the LoD).

4.8.3 Exploration of change in chemical concentrations over time

Summary data for mercury concentrations in dab muscle from samples analysed across the given period are given in Table 4.8.1.

To bring the data as close to a normal distribution as possible, the measured concentrations were converted into natural logarithm values (Ln) for the purpose of assessing trends. A plot of the overall change over time in mercury Ln concentrations in dab muscle from 2008 to 2019 is shown in Figure 4.8.1. The pattern for alternate years from 2011 reflect the biennial sampling of alternate sides of the country.

Table 4.8.1 Summary statistics for samples of mercury concentrations in dab muscle (mg/kg wet weight)[1]

Year	Number of stations	n	Mean	SD	Median	Min	Max	Q1	Q3
2008	23	111	0.11	0.078	0.080	0.010	0.37	0.050	0.15
2009	23	114	0.16	0.13	0.13	0.0050	1.1	0.073	0.21
2010	16	78	0.13	0.10	0.085	0.010	0.55	0.060	0.19
2011	15	74	0.077	0.033	0.070	0.030	0.17	0.060	0.090
2012	8	40	0.15	0.10	0.15	0.030	0.43	0.050	0.23
2013	15	73	0.10	0.043	0.090	0.040	0.26	0.070	0.11
2014	9	45	0.15	0.11	0.14	0.030	0.44	0.060	0.24
2015	14	66	0.10	0.042	0.090	0.040	0.23	0.060	0.12
2016	8	37	0.16	0.11	0.15	0.020	0.48	0.070	0.22

Year	Number of stations	n	Mean	SD	Median	Min	Max	Q1	Q3
2017	15	75	0.11	0.047	0.10	0.030	0.26	0.070	0.12
2018	9	45	0.13	0.094	0.11	0.029	0.36	0.047	0.18
2019	15	71	0.095	0.039	0.094	0.036	0.24	0.070	0.11

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Figure 4.8.1 Scatterplot of Ln mercury residues in the muscle of dab from marine waters around England between 2008 and 2019. Data shown are for individual samples. The solid black line shows the trend from a generalised additive model as a function of time (diagram courtesy of Cefas)
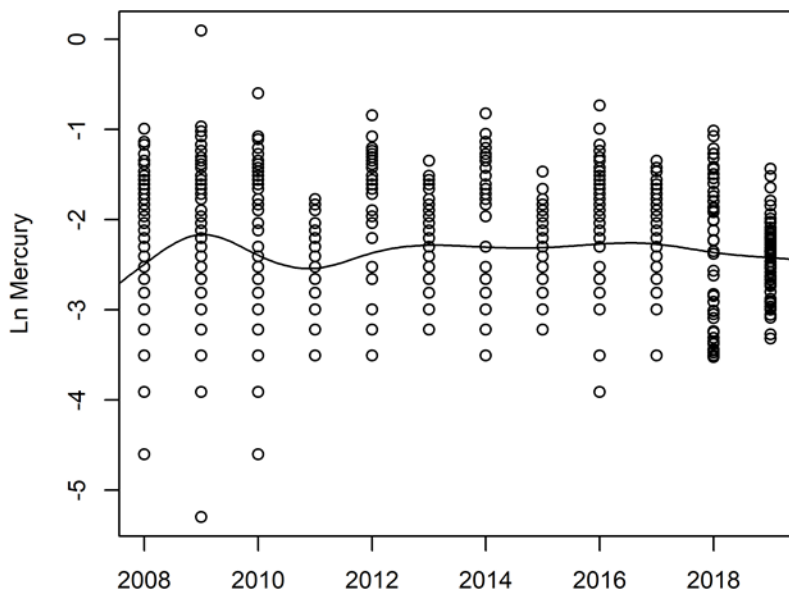
Minimum data requirements for trend assessment are met, though the monitoring regime has altered over time (see Section 4.8.1). For this reason, a trend assessment based on the whole time period has been made. A generalised additive model (GAM) (Wood, 2017) was used to estimate trend using the individual Ln sample data and time in years as the explanatory variable.

Whilst the trend line shown in Figure 4.8.1, based on all samples from all stations, is statistically significant (p <0.001) owing to the large number of observations, it appears to be very level, particularly from 2011 to 2019. The mean values in Table 4.8.1 also do not show any obvious trend.

To determine the overall trend for the dashboard, changes in concentrations over time at individual stations were assessed using the GAM method. This is because it is possible that analysing the results all together may give misleading conclusions for trends and mask intersite variations. Because there were sometimes only a few years' data for a station, the degrees of freedom for the smooth term in the GAM was set to the number of years minus 2 to allow for a successful fit.

When interpreting the GAM plots, two criteria were used: (1) the overall trend needed to be statistically significant at the 5% level and (2) the difference between the predicted values of the model in 2011 and 2019 needed to be statistically significant at the 5% level. For the second criterion, if the ratio of the model difference to the standard error of the model difference was greater than 2, then the difference was taken to be statistically significant at the 5% level. This approach was picked so the results were informed by the most-recent data where the total number of stations were increased and the sampling split over two years, because it covered a reasonable time period and because it maximised the available data for this period.

Scatterplots of results for samples taken at these stations can be found in Appendix B, Figure B.4.8.1. A statistic D was then calculated as follows: The number of stations showing downward trends was deducted from those showing upward ones. This value was reported as a percentage of the total number of stations examined. Where D ≥ 20%, an overall upward trend is assigned; where D ≤ –20%, a downward trend is reported.[6] Between these two values, a level result of no observed change is recorded.

The majority of trends in concentrations were either level or not statistically significant. Two stations appear to show a downward trend and two show an upward trend out of the 25 sites (see Appendix B, Figure B.4.8.1). These 4 stations are in the north. The resulting D value (D 0%) strongly supports the assignment of no observed change (↔) and this is used within the dashboard.

4.8.4 Thresholds

There are currently no agreed criteria for assessing the ecological significance of mercury concentrations under OSPAR. However, under the Water Framework Directive (Standards and Classification) Directions 2015 (UK Government, 2015), a biota EQS of 20µg/kg wet weight is available. This value is intended to protect predators from the effects of secondary poisoning and, therefore, is relevant to whole fish. Because the tissue measured in dab is muscle, the application of this EQS as a threshold for the indicator can be considered an interim and over-precautionary.

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[6] The use of the D statistic is a pragmatic approach for reflecting the national trend from statistically significant trends at stations. The thresholds of 20% are proposed because they give a balance between reporting trends when there is very little difference and not seeing any trends at all.

Data for 2018 and 2019 were compared against the EQS of 20μg/kg wet weight to assess the most recent results for all sites around the country. All 116 samples were above this value, equating to 100% exceedance of the threshold, and these are the results that are used in the dashboard.

4.9 Persistent, bioaccumulative and toxic substances in marine fish: polybrominated diphenyl ethers, polychlorinated biphenyls and perfluorooctanesulfonic acid

Polybrominated diphenyl ethers

Polychlorinated biphenyls

Perfluorooctanesulfonic acid

4.9.1 Data source

Data on concentrations of PBDEs, PCBs and PFOS in marine fish livers are available for dab (*Limanda limanda*). These data are collected as part of MSFD–OSPAR monitoring for assessing good environmental status. The data are collected and held by Cefas.

Data used in this assessment are also submitted, as part of the wider UK dataset, to the DOME (marine environment) data portal for the ICES (ICES, 2021a).

Sites are selected on the basis that they reliably support dab populations that can be sampled for analysis. There are a minimum of three sites within each OSPAR hydro-geographical sub-region (OSPAR Commission, 2020) and there are no direct impacts from local sources.

Data for PBDEs and PCBs have been provided for 2008–2019 covering 25 stations, although monitoring at one of these was only conducted in 2014. Between 2008 and 2010, sampling around the country was done annually and covered 16–20 stations each year. From 2011 onwards, fish were collected at east and west coast stations on alternate years. For these years, there are data for 12–15 east coast stations (odd years) and 8–9 west coast ones (even years).

Data for PFOS covers measurements from 2014, 2015, 2018, and 2019 taken at 25 coastal sampling stations. Again, fish were sampled from the east and west coast in alternate years. There are data for 9 west coast stations or 14–15 east coast ones each year. Most, but not all, sites had repeat visits.

All data relate to designated English waters, with the exception of those from a Welsh station in the Bristol Channel as this is a shared water body in which fish are likely to move freely across territorial waters.

Typically, 5 or fewer pools of fish were sampled around each station. Each pool comprised 5 fish.

4.9.2 Data structure

Data are available for 11 individual PBDE congeners – 17, 28, 47, 66, 85, 99, 100, 138, 153, 154, and 183 – and the summed concentrations of these congeners (SUM 11PBDE). All concentration data are reported in mg/kg wet weight and those for key congeners and the SUM 11PBDE value are also reported as their converted lipid weight values. The summed lipid weight values are summarised in Section 4.9.3 and results for individual PBDEs have been used for the threshold assessment (Section 4.9.4).

For PCBs, data are available for 25 individual PCB congeners. These include the ICES-7 PCB congeners: 28, 52, 101, 118, 138, 153, and 180. The remaining 18 PCB congeners for which there are data are 18, 31, 44, 47, 49, 66, 105, 110, 128, 141, 149, 151, 156, 158, 170, 183, 187, and 194. Data are also available for the summed concentrations of the ICES-7 PCBs (SUM ICES-7) and for the 25 congeners in total (SUM 25PCB). All concentration data are reported in mg/kg wet weight. The individual ICES-7 PCBs and both summed values are also reported as their converted lipid weight concentrations. The summed lipid weight values are summarised in Section 4.9.3. Results for the individual ICES-7 PCBs have been used for the threshold assessment (Section 4.9.4).

The LoDs for individual PBDE and PCB congeners varied within the datasets of 775 samples. However, only three samples had non-detects for all congeners: two samples for PBDEs in 2016 at one station and one sample for PCBs in 2015. The LoDs in these cases ranged from 0.00011 to 0.00012mg/kg wet weight. For the other samples, congener results reported below the LoD were assigned a zero value for the purposes of summing.

For PFOS, data are reported in µg/kg wet weight. All 214 samples analysed had concentrations that were above the LoD.

The data summaries that were provided consisted of results from individual pool samples taken around the stations. These individual samples were used in the trend and threshold assessments rather than station means for the purposes of the dashboard indicator. This is because the pooled samples are already representative of a mean of 5 fish and this approach allows assessment of trends across the stations and nationally.

4.9.3 Exploration of change in chemical concentrations over time

Summary data for SUM 11PBDE, SUM 25PCB and SUM ICES-7, and PFOS concentrations in dab liver across the analysed period are given in Tables 4.9.1, 4.9.2 and 4.9.3. To bring the data as close to a normal distribution as possible, the measured concentrations were converted into natural logarithm values (Ln) for the purpose of assessing trends.

Table 4.9.1 Summary statistics for samples of SUM 11PBDE concentrations in dab liver (mg/kg lipid weight)[1]

Year	Number of stations	n	Mean	SD	Median	Min	Max	Q1	Q3
2008	20	94	0.069	0.061	0.044	0.0020	0.22	0.020	0.10
2009	19	94	0.028	0.022	0.023	0.0014	0.14	0.012	0.036
2010	16	79	0.050	0.055	0.033	0.0012	0.35	0.022	0.063
2011	12	60	0.040	0.037	0.034	0.0028	0.16	0.013	0.052
2012	8	40	0.036	0.028	0.032	0.0033	0.10	0.010	0.056
2013	15	73	0.043	0.042	0.032	0.0019	0.19	0.015	0.050
2014	9	44	0.025	0.025	0.020	0.0019	0.12	0.0063	0.036
2015	14	66	0.033	0.045	0.019	0.00037	0.22	0.0072	0.030
2016	8	37	0.016	0.012	0.012	0	0.044	0.0085	0.022
2017	15	72	0.023	0.026	0.016	0.0014	0.14	0.0081	0.028
2018	9	45	0.010	0.0069	0.0075	0.0014	0.026	0.0048	0.018
2019	15	71	0.023	0.028	0.013	0.0012	0.14	0.0078	0.024

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Table 4.9.2 Summary statistics for samples of SUM 25PCB and SUM ICES-7 concentrations in dab liver (mg/kg lipid weight)[1]

Year	Number of stations	n	SUM	Mean	SD	Median	Min	Max	Q1	Q3
2008	20	94	25PCBs	0.81	0.75	0.56	0.17	3.9	0.30	1.1
2008	20	94	ICES-7	0.51	0.46	0.35	0.12	2.4	0.21	0.65
2009	19	94	25PCBs	0.49	0.41	0.36	0.049	1.9	0.18	0.68
2009	19	94	ICES-7	0.31	0.24	0.24	0.049	1.1	0.14	0.38
2010	16	79	25PCBs	0.66	0.64	0.44	0.10	3.5	0.26	0.76
2010	16	79	ICES-7	0.43	0.43	0.29	0.067	2.6	0.19	0.54
2011	12	60	25PCBs	0.39	0.22	0.31	0.14	1.1	0.24	0.44
2011	12	60	ICES-7	0.26	0.15	0.21	0.092	0.77	0.17	0.28
2012	8	40	25PCBs	0.88	0.65	0.73	0.15	2.6	0.41	1.13
2012	8	40	ICES-7	0.53	0.40	0.44	0.090	1.7	0.24	0.69
2013	15	73	25PCBs	0.43	0.23	0.37	0.13	1.4	0.27	0.51
2013	15	73	ICES-7	0.30	0.18	0.23	0.11	0.97	0.18	0.33
2014	9	44	25PCBs	0.75	0.68	0.56	0.10	3.2	0.27	0.99
2014	9	44	ICES-7	0.45	0.41	0.32	0.062	1.9	0.16	0.61
2015	14	66	25PCBs	0.38	0.31	0.26	0	1.6	0.21	0.41
2015	14	66	ICES-7	0.26	0.21	0.18	0	1.1	0.14	0.26
2016	8	37	25PCBs	0.50	0.35	0.40	0.12	1.8	0.28	0.65
2016	8	37	ICES-7	0.29	0.21	0.23	0.069	1.1	0.16	0.37

Year	Number of stations	n	SUM	Mean	SD	Median	Min	Max	Q1	Q3
2017	15	72	25PCBs	0.32	0.18	0.27	0.13	1.1	0.21	0.34
2017	15	72	ICES-7	0.22	0.13	0.19	0.090	0.79	0.15	0.24
2018	9	45	25PCBs	0.39	0.23	0.30	0.12	0.94	0.21	0.56
2018	9	45	ICES-7	0.23	0.14	0.18	0.080	0.57	0.12	0.31
2019	15	71	25PCBs	0.29	0.17	0.23	0.070	0.78	0.18	0.36
2019	15	71	ICES-7	0.20	0.11	0.16	0.042	0.53	0.13	0.25

[1]n: number of samples analysed; SUM: summed values; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Table 4.9.3 Summary statistics for samples of perfluorooctanesulfonic acid concentrations in dab liver (µg/kg wet weight)[1]

Year	Number of stations	n	Mean	SD	Median	Min	Max	Q1	Q3
2014	9	34	6.8	3.8	6.6	1.3	18	3.9	8.5
2015	14	64	4.1	2.7	3.6	0.51	13	2.2	5.6
2018	9	45	7.9	4.5	7.8	1.4	18	4.4	9.9
2019	15	71	2.9	1.9	2.6	0.65	12	1.5	3.9

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Plots of the overall change over time in Ln SUM 11PBDE and SUM25PCB concentrations in dab liver from 2008 to 2019 are shown in Figure 4.9.1.

Figure 4.9.1 Scatterplots of Ln SUM PBDE and SUM 25PCB residues in the liver of dab from marine waters around England between 2008 and 2019. Data shown are for individual samples. The solid black line shows the trend from a generalised additive model as a function of time (diagram courtesy of Cefas)
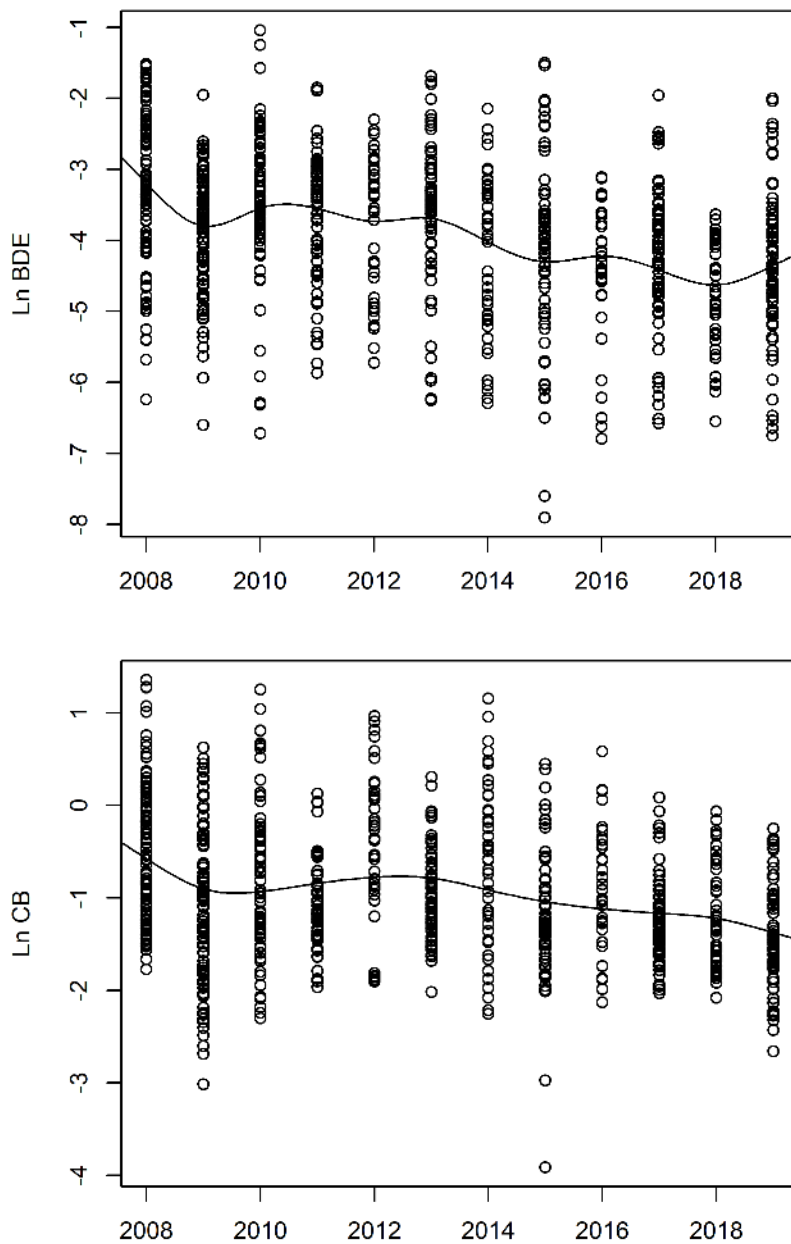
Minimum data requirements for trend assessment are met for both PBDEs and PCBs, though the monitoring regime has altered over time (see Section 4.9.1). A GAM (Wood, 2017) was used to estimate trends using the individual Ln sample data for both groups of substances and time in years as the explanatory variable.

Based on all samples for all stations, the SUM PBDE plot in Figure 4.9.1 shows a clear downward trend over time; the p-value for the overall trend line is strongly statistically significant (p <0.001). The waviness in the modelled trend line from 2011 onwards may reflect the slightly lower concentrations seen at the west coast stations compared with

those from the east coast. The downward trend in both areas over this time is also reflected in the median values in Table 4.9.1.

For SUM 25PCBs, the p-value for the overall trend line (Figure 4.9.1) indicates a strong statistically significant trend (p <0.001).

To determine the overall trends for these groups of substances for the dashboard, temporal changes in concentrations at individual stations were assessed. This is because it is possible that analysing the results all together may give misleading conclusions for trends and mask intersite variations. Scatterplots of results for samples taken at the stations can be found in Appendix B, Figures B.4.9.1 and B.4.9.2 for SUM 11 PBDEs and SUM 25PCBs, respectively.

A GAM was used again for the assessment. The same criteria for interpreting the GAM plots for individual stations as described in Section 4.8.3 was used here. The statistic D for SUM 11PBDEs and SUM 25PCBs was then calculated from the station results as also described in Section 4.8.3.

For SUM 11PBDEs, 11 east coast and 6 west coast stations show downward trends out of the 25 sites (see Appendix B, Figure B.4.9.1). No stations exhibited any upward trends. Therefore the resulting D value (–68%) strongly supported the assignment of a downward trend (↓) and this is used within the dashboard.

The individual stations around which SUM 25PCBs in dab liver were measured appear to be going down overall, although the situation is less distinct than for the PBDEs. Particularly on the east coast, there are a lot of stations for which the trend is approximately level or where there is not a confirmed statistical trend. There are 7 downward trends – 3 in the east and 4 in the west – and 2 upward trends split evenly geographically at the 25 stations assessed (see Appendix B, Figure B.4.9.2). The resulting D value of –20% suggests a downward trend (↓) and this assignment is used within the dashboard.

For PFOS, there are limited data to make temporal comparisons. Fourteen stations on the east coast and 6 on the west have more than one sample for both available years' of data (2015/2019 and 2014/2018, respectively). The individual sample results per station for those years are shown in Appendix B, Figure B.4.9.3. However, because of the low number of samples, the statistical power to detect change is low.

A standard t-test was used to compare the mean levels of PFOS across the corresponding two years of data for each side of the country. This was used because the data were reasonably symmetric and the variation within years was similar for each station. The t-test also assumes that the data are independent within stations, reflecting that each sample comes from different fish. A two-sided t-test was used as there was no theoretical information as to whether the trend was up or down.

Four of the east coast stations show statistically significant differences: 2 upward trends and 2 down. For the west coast, 2 upward trends were observed. The corresponding D

value is 10% indicating no observed change. However, while statistically significant differences were seen, the minimum data requirements are not met for reporting a trend for this substance in dab livers and so the trend assessment information is left blank in the dashboard.

4.9.4 Thresholds

There are no EACs derived for PBDE congeners under OSPAR. However, an approach recommended by the OSPAR MIME (ICES, 2021b) is to use the Canadian FEQGs for biota (Environment and Climate Change Canada, 2013). This approach results in threshold values for 6 of the 11 individual PBDEs (Table 4.9.4).

The FEQGs for biota either relate to fish health (concentrations that should not cause adverse effects on fish) or mammalian wildlife health (concentrations in fish that should not cause adverse effects on mammalian predators). As such, they might be less protective than an EAC (the concentration that should not cause adverse effects on the most sensitive marine organisms). Conversely, the FEQGs are for whole fish concentrations which will be lower values than for the liver concentrations to which they are generally compared. To make the data directly comparable in the latter case, we have compared the lipid weight values of PBDEs in fish liver against the corresponding lipid weight thresholds so that they are assessed on the same basis.

The EAC thresholds used in OSPAR assessments for the individual ICES-7 PCB congeners are used here (Table 4.9.4).

The number of individual samples that had concentrations above these proposed values was calculated for the most recent pair of years, that is 2018 and 2019, to allow assessment of results taken from across the whole country.

Table 4.9.4 Suggested values that can be used as dashboard thresholds for polybrominated diphenyl ethers and polychlorinated biphenyls

PBDE congener number	Canadian FEQG based values (µg/kg lipid weight)[1]	Canadian FEQG based values (µg/kg wet weight)	PCB congener number	OSPAR EAC (µg/kg lipid weight)
28	2400	120	**28**	67
47	880	44	**52**	108
99	20	1	**101**	121
100	20	1	**118**	25

PBDE congener number	Canadian FEQG based values (μg/kg lipid weight)[1]	Canadian FEQG based values (μg/kg wet weight)	PCB congener number	OSPAR EAC (μg/kg lipid weight)
153	80	4	**138**	317
154	80	4	**153**	1585
–	–	–	**180**	469

[1] Converted from wet weight threshold using a conversion factor of 20 (for a standard whole fish with a lipid content of 5% (EC, 2014)).

For PBDEs, 2 samples around 1 station on the east coast in 2019 out of 116 samples in total (1.7%) had PBDE100 concentrations in dab liver that exceeded the FEQG. There were no exceedances for the other PBDE congeners for which Canadian FEQGs are available.

For PCBs, 24 samples from the east coast in 2019 and 15 from the west in 2018 out of 116 in total (34%) had mean PCB118 concentrations in dab liver that exceeded the EAC. There were no exceedances for the other ICES-7 PCB congeners for which OSPAR EACs are available.

An EAC is not available for PFOS, but a $QS_{sec\ pois}$ has been derived through the EU EQS derivation process, which considers different protection goals. The $QS_{sec\ pois}$ does not have statutory status as an EQS because it is not the most critical (lowest) QS. The EQS has a different protection goal of human health; however the $QS_{sec\ pois}$ is the most appropriate to use here.

The derived $QS_{sec\ pois}$ for PFOS is 33μg/kg wet weight (EC, 2011b). However, it should be noted that the threshold is based on whole fish concentrations whereas the available data for marine fish are for concentrations of PFOS in liver, which are typically 3 times higher than for whole fish. Caution is therefore needed in interpreting this threshold assessment in terms of potential risk to predators as the result is likely to be over-precautionary. None of the 116 samples exceeded this value.

The values used for the dashboard indicator are the percentage of samples that exceeded the above-mentioned thresholds for 2018 and 2019 combined.

4.10 Persistent, bioaccumulative and toxic substances in marine mammals: polybrominated diphenyl ethers, polychlorinated biphenyls and perfluorooctanesulfonic acid

Polybrominated diphenyl ethers

Polychlorinated biphenyls

Perfluorooctanesulfonic acid

4.10.1 Data source

Concentration data on PBDEs and PCBs in blubber and PFOS in liver samples are available for harbour porpoise (*Phocoena phocoena*). These data are collected as part of the UK Cetacean Strandings Investigation Programme (CSIP) under a Service Level Agreement between Defra and Cefas. The first samples were analysed in 1992 and a minimum of 20 UK samples are analysed each year.

Tissue samples are taken opportunistically from marine mammal stranding and bycatch incidents. A subset is chosen for analysis that contains an even split of bycatch vs stranding, adult vs juvenile and male vs female, with samples covering England, Wales and Scotland. The contaminant data are collected and held by Cefas.

Data for individual animals collected from the UK CSIP sites were provided for the years 2004 to 2018 (excluding 2009 for PBDEs) for PBDEs and PCBs and for 2001–2003 and 2012–2018 for PFOS for use in the current report.

Data were not restricted to the England level, as for most of the other metrics within this indicator. This is because the wider geographical dataset maintains an even split between animal types in the dataset – not biasing a certain type that may be more (or less) susceptible to accumulating contaminants, such as adult males or those dying from infectious disease or starvation. It also reflects that these species are likely to move more widely along the UK coast and enables more robust trend determination because of the increased number of samples per year. This approach is consistent with other marine indicators within the 25-YEP Outcome Indicator Framework.

4.10.2 Data structure

Each sample is from a single individual. The PBDE and PCB congeners were the same as those described in Section 4.9.2, that is PBDEs 17, 28, 47, 66, 85, 99, 100, 138, 153, 154, and 183, and PCBs 18, 28, 31, 44, 47, 49, 52, 66, 101, 105, 110, 118, 128, 138, 141, 149, 151, 153, 156, 158, 170, 180, 183, 187, and 194.

SUM 11PBDE and SUM 25PCB data are reported in mg/kg wet weight and have been normalised to lipid weight to enable comparison with potential threshold concentrations. PFOS concentration data are reported in µg/kg wet weight.

The LoDs for individual PBDE and PCB congeners varied within the datasets of 380 and 372 samples, respectively. Generally, the PBDE congeners had LoDs of approximately 0.002mg/kg wet weight, though samples in 2018 had values reported below 0.001mg/kg wet weight. Those for PCB congeners were around 0.01mg/kg wet weight. When concentrations of individual PBDE and PCB congeners were below the LoD, they were assigned a value of zero when summing concentrations to calculate SUM 11PBDE and SUM 25PCB values.

None of the 180 samples analysed had PFOS concentrations below the LoD, except for 2 taken in 2002 (<32µg/kg wet weight); the detection limit has lowered since that date. These two values were assigned a value equal to half the LoD (16µg/kg wet weight) for statistical analysis.

4.10.3 Exploration of change in chemical concentrations over time

The distribution of data by year is summarised in Tables 4.10.1, 4.10.2 and 4.10.3 for SUM 11PBDE, SUM 25PCB and PFOS concentrations, respectively, in harbour porpoise. To bring the data as close to a normal distribution as possible, these concentrations were converted into natural logarithm values (Ln) for the purpose of assessing trends.

Table 4.10.1 Summary statistics for samples of SUM PBDE concentrations in harbour porpoise blubber (mg/kg lipid weight)[1]

Year	n	Mean	SD	Median	Min	Max	Q1	Q3
2004	35	1.11	1.14	0.828	0.276	5.99	0.396	1.31
2005	55	0.636	0.643	0.517	0.0314	4.01	0.227	0.797
2006	41	0.656	0.634	0.456	0.0983	3.15	0.228	0.799
2007	39	0.461	0.460	0.311	0.0953	2.81	0.199	0.595
2008	30	0.385	0.370	0.240	0.0569	1.40	0.145	0.455

Year	n	Mean	SD	Median	Min	Max	Q1	Q3
2010	20	0.361	0.273	0.317	0.0581	1.21	0.145	0.482
2011	20	0.401	0.284	0.303	0.0880	1.19	0.209	0.545
2012	20	0.316	0.303	0.186	0.0576	1.34	0.136	0.373
2013	20	0.272	0.187	0.241	0.0297	0.602	0.132	0.382
2014	18	0.159	0.165	0.129	0.0143	0.738	0.0593	0.209
2015	20	0.223	0.218	0.133	0.0360	0.775	0.0882	0.287
2016	21	0.278	0.303	0.163	0.0630	1.43	0.114	0.369
2017	20	0.277	0.320	0.212	0.0119	1.48	0.134	0.288
2018	21	0.284	0.281	0.192	0.0293	1.12	0.117	0.330

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Table 4.10.2 Summary statistics for samples of SUM 25PCB concentrations in harbour porpoise blubber (mg/kg lipid weight)[1]

Year	n	Mean	SD	Median	Min	Max	Q1	Q3
2004	31	14.3	11.7	12.6	1.62	53.8	5.60	17.3
2005	49	15.0	13.4	11.1	1.03	60.2	5.37	20.5
2006	26	20.7	30.9	9.85	3.19	139	5.73	19.0
2007	28	11.8	11.5	7.87	1.75	44.8	3.50	14.7
2008	25	10.7	8.62	8.45	1.67	38.7	5.43	12.4
2009	23	18.1	23.8	7.79	0.662	81.2	3.94	17.2

Year	n	Mean	SD	Median	Min	Max	Q1	Q3
2010	20	11.5	11.1	7.33	0.693	36.6	3.59	16.7
2011	21	14.0	11.9	10.9	1.13	40.9	5.45	18.9
2012	22	16.0	23.5	8.47	1.13	108	3.99	17.5
2013	22	11.6	9.04	9.54	0.800	30.9	3.50	16.3
2014	23	18.7	21.8	10.7	1.35	103	5.53	23.3
2015	20	7.52	8.18	5.05	1.45	39.0	3.29	9.06
2016	21	20.1	39.6	6.68	1.23	181	3.16	13.0
2017	20	13.9	16.6	7.42	0.374	62.9	2.35	19.3
2018	21	17.1	18.6	11.5	1.21	67.0	3.25	23.6

[1]n: number of samples analysed; SUM: summed values; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Table 4.10.3 Summary statistics for samples of perfluorooctanesulfonic acid concentrations in harbour porpoise liver (µg/kg wet weight)[1]

Year	n	Mean	SD	Median	Min	Max	Q1	Q3
2001	10	873	629	952	138	1810	280	1305
2002	16	510	654	232	16.0	2420	120	616
2003	14	510	551	256	83.0	1820	161	691
2012	19	143	112	102	6.56	378	52.7	223
2013	20	190	157	142	26.3	533	77.7	248
2014	21	194	234	157	15.9	1144	73.2	196

Year	n	Mean	SD	Median	Min	Max	Q1	Q3
2015	20	170	157	116	12.8	615	69.9	241
2016	21	89.1	73.4	71.9	7.34	263	41.3	142
2017	21	95.5	76.6	75.7	15.2	292	47.2	118
2018	18	180	144	127	4.01	486	91.1	226

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Plots of the overall change in Ln SUM 11PBDE and SUM 25PCB concentrations in harbour porpoise blubber from 2004 to 2018 are shown in Figures 4.10.1 and 4.10.2, respectively.

Figure 4.10.1 Scatterplots of Ln SUM PBDE residues in the blubber of harbour porpoise from marine waters around the UK between 2004–2008 and 2010–2018. Data shown are for individual samples. The solid black line shows the trend from a generalised additive model as a function of time, the hashed red line shows the Ln value of the threshold given in Section 4.10.4 (diagram courtesy of Cefas)
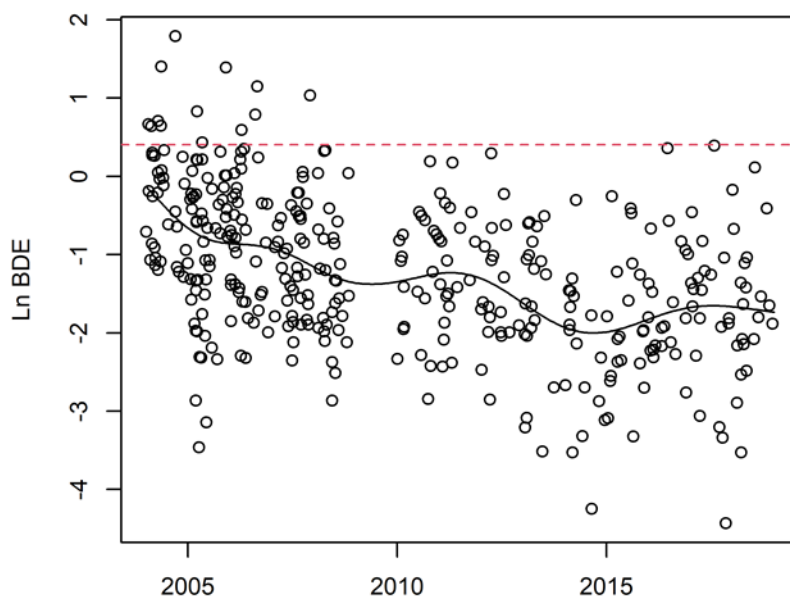
Figure 4.10.2 Scatterplots of Ln SUM 25PCB residues in the blubber of harbour porpoise from marine waters around the UK between 2004 and 2018. Data shown are for individual samples. The solid black line shows the trend from a generalised additive model as a function of time, the hashed red line and the dotted blue line show the Ln values of the lower and upper thresholds, respectively, given in Section 4.10.4 (diagram courtesy of Cefas)
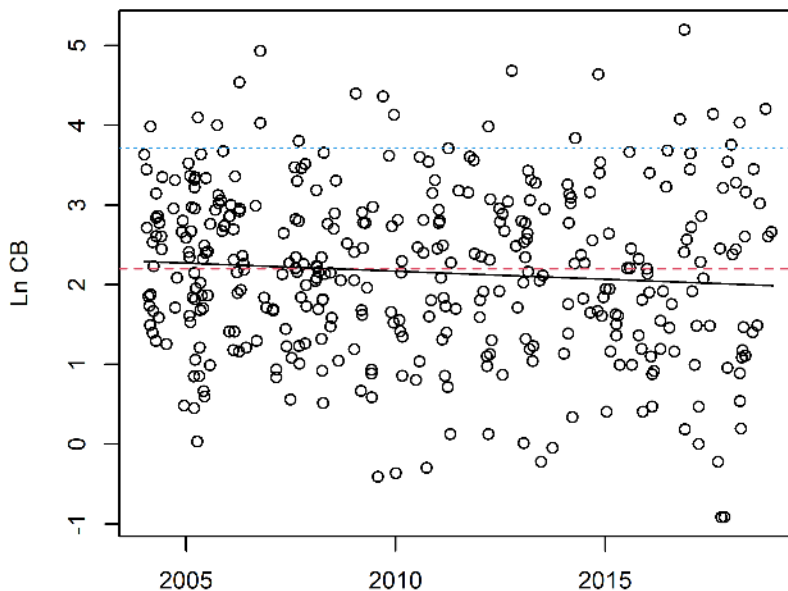
Minimum data requirements for trend assessment are met for all substances. A GAM ([Wood, 2017](#)) was used to estimate trends using the individual Ln sample data for SUM 11PBDE, SUM 25PCB and PFOS and time in years as the explanatory variable. Statistically significant trends were those at the 5% level.

For PBDEs, there is a clear downward trend over time, although the means (Table 4.10.1) and the plot in Figure 4.10.1 suggests some levelling off in more-recent years. The p-value for the overall trend line is strongly statistically significant (p <0.001) and the assignment of a downward trend (↓) is used in the dashboard.

For PCBs, while the GAM line shown in Figure 4.10.2 suggests a marginal downward trend, it is not statistically significant (p = 0.1). Therefore the assignment of no change in concentrations (↔) is given in the dashboard.

For PFOS, the trends have been looked at in two ways: assessment of all data from the two periods 2001–2003 and 2012–2018 and closer assessment of the latter range of years. Scatterplots of these data are shown in Figure 4.10.3.

Figure 4.10.3 Scatterplots of Ln perfluorooctanesulfonic acid residues in the liver of harbour porpoise from marine waters around the UK between 2001–2003 and 2012–2018 (top) and 2012–2018 (bottom). Data shown are for individual samples. The solid black line shows the trend from a generalised additive model as a function of time, the hashed red lines show the Ln value of the threshold given in Section 4.10.4 (diagram courtesy of Cefas)
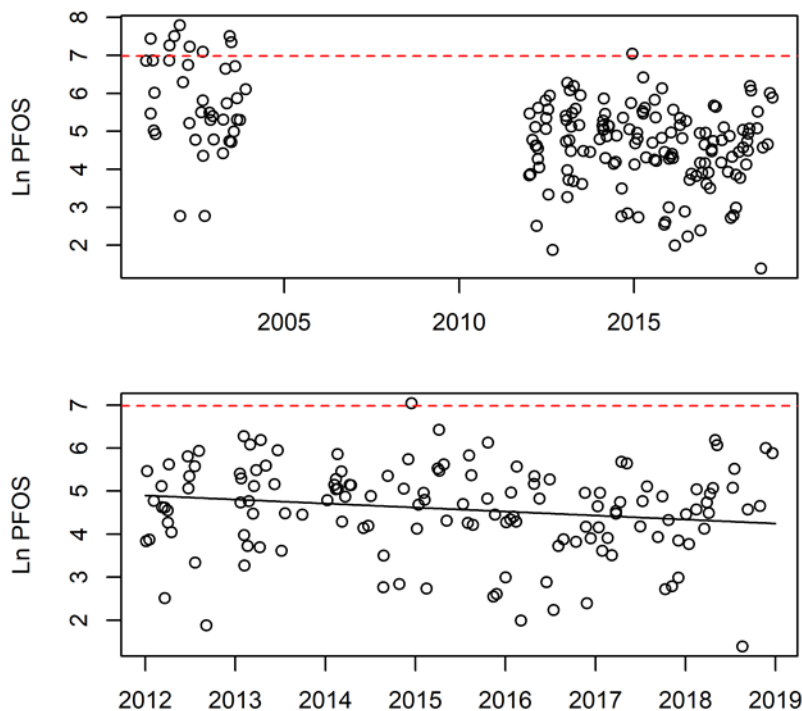
The measured means of the two groups of years of PFOS data, shown as Ln values in the top plot of Figure 4.10.3, were 600µg/kg wet weight for 2001–2003 and 151µg/kg wet weight for 2012–2018. A t-test confirmed that there was a strong statistically significant difference (p <0.001) between the two mean values for those periods.

The bottom plot in Figure 4.10.3 shows the results of a GAM fitted to the 2012–2018 Ln PFOS data. This shows a strong statistically significant linear decrease with time (p <0.001). The summary data in Table 4.10.3 suggest a possible upturn in 2018, but data from future years would be needed to see if this continued.

Overall, PFOS residues in harbour porpoise liver appear to show a downward trend and the corresponding assignment (↓) is given in the dashboard.

4.10.4 Thresholds

There are no established statutory or international thresholds for SUM 11PBDE and SUM 25PCB concentrations in blubber.

A potential threshold of 1.5mg/kg lipid weight has been proposed for SUM 11PBDEs in blubber (Hall et al., 2003) based on thyroid disruption in juvenile grey seals (*Halichoerus grypus*). None of the 21 individuals analysed in 2018 had blubber SUM 11PBDE concentrations exceeding this proposed value. The proportion of harbour porpoise with

blubber SUM 11PBDE concentrations >1.5mg/kg lipid weight in 2018 (0%) is used as the dashboard entry.

For SUM 25PCBs, two potential thresholds have been proposed: 9mg/kg lipid weight for immunological effects in aquatic mammals (Kannan et al., 2000) and 41mg/kg lipid weight for reproductive effects, based on studies on seals (Helle et al., 1976). Twelve individuals exceed the lower threshold and 3 of these the upper value out of 21 samples from 2018. The dashboard entry is based on the proportion of harbour porpoise analysed in 2018 with blubber SUM 25PCB concentrations that exceeded the proposed threshold for immunological effects (57%).

There is no threshold established for PFOS concentrations in marine mammal liver tissue but Lam et al. (2016) suggested a tentative critical concentration in cetacean (dolphin) liver of 775µg/kg wet weight for PFOS. This was based on toxicological information – a no observed adverse effects level – for mammalian species (rat) with an assessment factor added to account for cross-species extrapolation. This value has been converted into a threshold of 1075µg/kg wet weight for harbour porpoise liver taking into account the mass of these animals. There were no exceedances of this value out of 18 samples analysed in 2018. The dashboard entry indicates the proportion of harbour porpoise that had liver concentrations that exceeded this tentative critical concentration in 2018 (0%).

4.11 Heavy metals in sparrowhawk (*Accipter nisus*): lead and cadmium

Lead

Cadmium

4.11.1 Data source

Data on lead and cadmium in sparrowhawk livers are provided by the PBMS (UKCEH, 2020). UKCEH has reported on concentrations of lead in liver in a series of reports (Walker et al., 2011, 2013, 2014). Data for cadmium concentrations in liver have usually been obtained at the same time as those for lead because they are determined through a common mode of chemical analysis, but the data have not been published to date.

Livers were excised from individual sparrowhawks found dead throughout England. Most animals died as a result of collisions or starvation, but some birds have died from other causes. Data on concentrations of lead and cadmium in liver are available for the period from 2007 to 2014, although data are not available for cadmium every year.

The data used for the dashboard are drawn from all birds collected and analysed for lead.

For cadmium, data restricted to first-year birds was used for temporal trend analysis. First years, defined as individuals hatched in the current or previous year to that in which they were found dead, were used because they are likely to provide a more-sensitive measure of annual change in exposure than adults, which may bioaccumulate cadmium over multiple years. This was not done for lead because half-lives for lead in liver are relatively short (1–3 months) (Krone, 2018). For the cadmium assessment of threshold exceedance, all birds were used irrespective of age.

4.11.2 Data structure

The data consist of measurements of lead and cadmium residues in the liver of a variable number of individuals that died each year between 2007 and 2014 for lead and in the years 2008 and 2010–2014 for cadmium. Data are reported as mg/kg dry weight for both elements. The LoD was 0.07mg/kg dry weight for lead and 0.01mg/kg dry weight for cadmium; values below the LoD were assigned values equal to half the LoD.

4.11.3 Exploration of change in chemical concentrations over time

The distribution of data by year is shown in Figure 4.11.1 and is summarised in Table 4.11.1.

Initial investigation of time trends for lead and cadmium concentrations in liver was conducted on log10-transformed values as this allowed the underlying assumptions of the statistical models to be met.

Figure 4.11.1 Scatterplots of lead (Pb) and cadmium (Cd) residues in the liver of all and first-year sparrowhawks, respectively, from England. Data shown are for individuals. Horizontal lines within plots indicate median values (diagram courtesy of UKCEH)
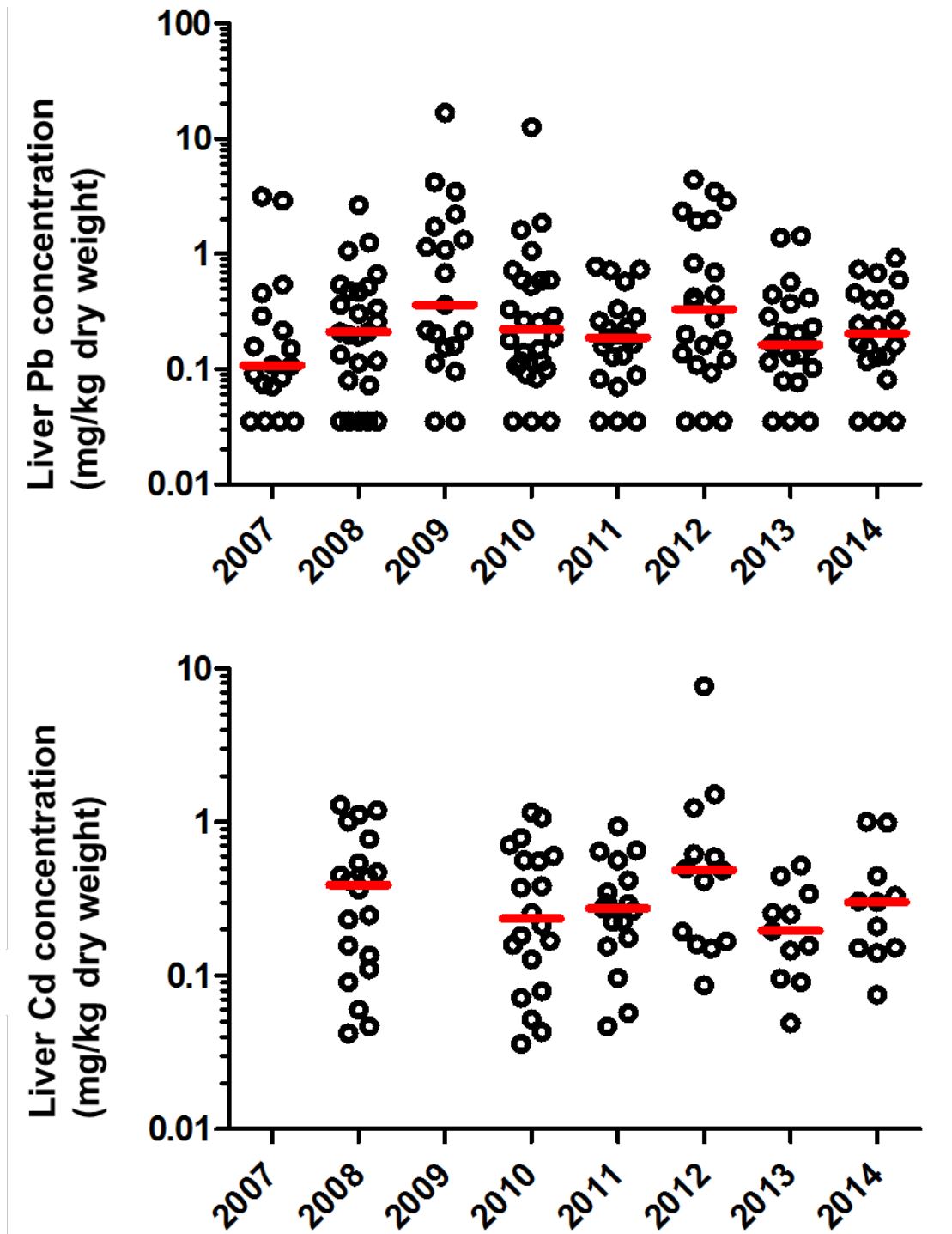
Table 4.11.1 Summary statistics for lead and cadmium concentrations in the liver of all and first-year sparrowhawks, respectively (mg/kg dry weight)[1]

Substance	Year	n	Mean	SD	Median	Min	Max	Q1	Q3
Lead	2007	18	0.473	0.938	0.108	0.035	3.15	0.062	0.330
Lead	2008	26	0.400	0.554	0.209	0.035	2.66	0.078	0.487
Lead	2009	17	1.94	4.03	0.363	0.035	16.8	0.157	1.97
Lead	2010	25	0.901	2.48	0.221	0.035	12.6	0.103	0.595
Lead	2011	22	0.257	0.232	0.188	0.035	0.782	0.087	0.296
Lead	2012	22	0.961	1.28	0.331	0.035	4.40	0.118	1.930
Lead	2013	22	0.308	0.384	0.163	0.035	1.43	0.096	0.381
Lead	2014	20	0.300	0.259	0.204	0.035	0.924	0.121	0.441
Cadmium	2008	20	0.460	0.406	0.390	0.042	1.29	0.116	0.718
Cadmium	2010	20	0.380	0.344	0.235	0.036	1.16	0.091	0.593
Cadmium	2011	17	0.334	0.242	0.275	0.047	0.942	0.166	0.491
Cadmium	2012	13	1.06	2.04	0.485	0.087	7.71	0.164	0.928
Cadmium	2013	11	0.232	0.151	0.196	0.049	0.519	0.096	0.342
Cadmium	2014	11	0.374	0.327	0.301	0.075	1.01	0.152	0.445

[1]n: number of individuals analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

There was no observed short-term (2009–2014) or longer term (2007–2014) temporal trend for lead residues in liver (F <3.7, p >0.05). For cadmium residues in liver of juvenile birds there was also no statistically significant trend (F <1.11, p = 0.36) from 2008 to 2014. Therefore, the assignment of no change in concentrations (↔) is given in the dashboard.

4.11.4 Thresholds

Thresholds are available in the literature for lead and cadmium in birds though these are based on a limited datasets.

Concentrations of lead in liver of >6mg/kg wet weight are associated with clinical poisoning in individual Falconiformes (Fransome and Pain, 2011). Using a mean wet weight to dry weight conversion factor for sparrowhawks of 3.52 (± 0.02; n = 1454)[7], a concentration of 6mg/kg wet weight is the equivalent of a dry weight concentration of 21mg/kg. This is the threshold proposed for use in the dashboard assessment.

A cadmium residue of 45–70mg/kg wet weight in liver has been suggested for adult full-grown birds, the exceedance of which may be associated with adverse physiological effects including alterations to energy metabolism or structural/functional damage to kidneys, testes, liver, gut, or salt glands (Wayland and Scheuhammer, 2011). The threshold residue for growing birds has not been defined but may be lower. Applying the wet weight to dry weight conversion factor of 3.52 as above, 45–70mg/kg wet weight is equivalent approximately to 160–250mg/kg dry weight. The lower value of this range (160mg/kg dry weight) is the suggested threshold for the dashboard.

Data for all birds used for the threshold assessments of lead and cadmium residues in sparrowhawk liver are given in Tables 4.11.1 and 4.11.2, respectively.

Table 4.11.2 Summary statistics for cadmium concentrations in the liver of all sparrowhawks (mg/kg dry weight)[1]

Year	n	Mean	SD	Median	Min	Max	Q1	Q3
2008	24	0.525	0.417	0.444	0.042	1.430	0.141	0.739
2010	25	0.481	0.424	0.402	0.036	1.650	0.144	0.749
2011	22	0.357	0.235	0.285	0.047	0.942	0.204	0.584
2012	22	1.870	2.966	0.571	0.045	11.30	0.187	1.555
2013	22	0.585	0.720	0.312	0.049	2.710	0.186	0.615

[7] R. F. Shore unpublished data; mean (± standard error) of 3.52±0.02 based on measurements on 1454 livers.

Year	n	Mean	SD	Median	Min	Max	Q1	Q3
2014	20	0.569	0.445	0.412	0.075	1.800	0.233	0.947

[1]n: number of individuals analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

None of the sparrowhawks for which data are available exceeded the thresholds for lead or cadmium in liver. Results for 2014 are incorporated into the dashboard.

4.12 Heavy metals in sparrowhawk (*Accipter nisus*): nickel, copper and zinc
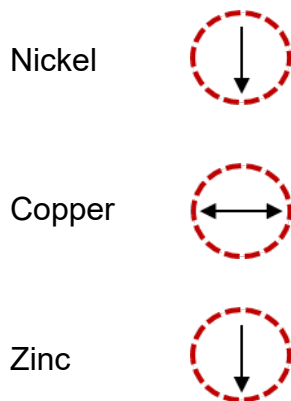
Nickel

Copper

Zinc

4.12.1 Data source

Data on nickel, copper and zinc in sparrowhawk livers is provided by the PBMS (UKCEH, 2020) which has quantified these metals but not published reports.

Livers were excised from individual sparrowhawks found dead throughout England. Most animals died as a result of collisions or starvation, but some birds have died from other causes. Data on concentrations of nickel, copper and zinc in liver are available for the period from 2007 to 2014.

The data used for the dashboard are drawn from all birds collected and analysed. These are essential metals and, therefore, the concentrations in liver should be regulated and there should not be any age-related bioaccumulation of these metals.

4.12.2 Data structure

The data consist of measurements of nickel, copper and zinc in the liver of a variable number of individuals that died each year between 2007 and 2014. Data are reported as mg/kg dry weight for all three metals.

A common LoD of 0.022mg/kg dry weight was applied for nickel. This resulted in 17 samples assigned as non-detected for the statistical analysis even though they were detected originally. This approach reduced the impact among-batch variation in LoDs had on the statistical analysis. Values below the LoD for nickel were assigned values equal to half the LoD. For copper and zinc, all measurements were above the limit of detection.

4.12.3 Exploration of change in chemical concentrations over time

The distribution of data by year for each of nickel, copper and zinc is shown in Figure 4.12.1 and is summarised in Tables 4.12.1, 4.12.2 and 4.12.3, respectively. For nickel, the dataset was dominated by samples that had concentrations below the LoD with only 16.5% of samples being above this value.

Figure 4.12.1 Scatterplots of nickel (Ni), copper (Cu) and zinc (Zn) residues in the liver of sparrowhawks from England. Data shown are for individuals. Horizontal lines within plots indicate median values (diagram courtesy of UKCEH)
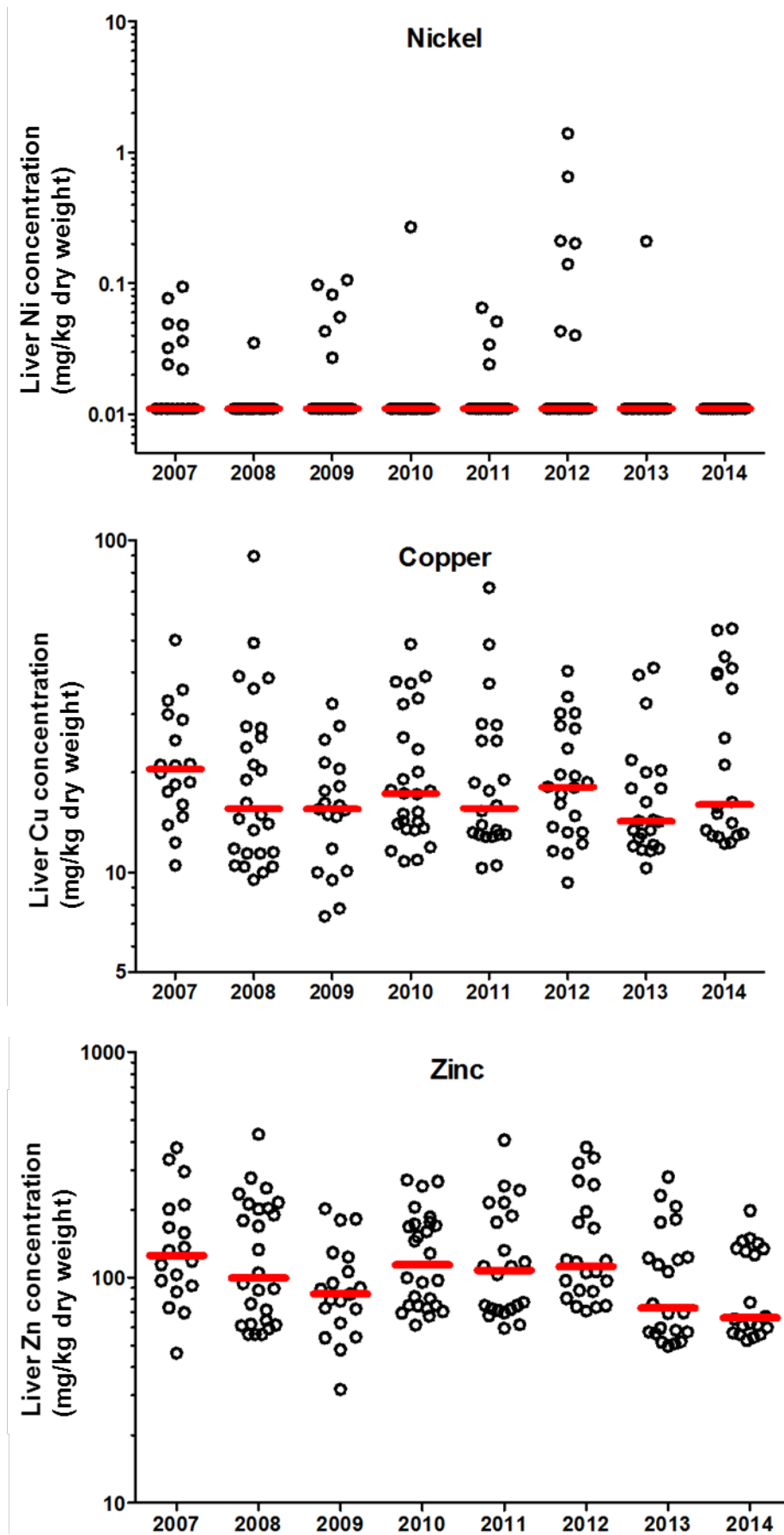
Table 4.12.1 Summary statistics for nickel concentrations in sparrowhawk liver (mg/kg dry weight)[1]

Year	n	Mean	SD	Median	Min	Max	Q1	Q3
2007	18	0.027	0.025	0.011	0.011	0.094	0.011	0.039
2008	26	0.012	0.005	0.011	0.011	0.035	0.011	0.011
2009	19	0.029	0.032	0.011	0.011	0.106	0.011	0.043
2010	26	0.021	0.051	0.011	0.011	0.269	0.011	0.011
2011	23	0.017	0.014	0.011	0.011	0.065	0.011	0.011
2012	21	0.135	0.323	0.011	0.011	1.39	0.011	0.092
2013	22	0.020	0.042	0.011	0.011	0.209	0.011	0.011
2014	20	0.011	0	0.011	0.011	0.011	0.011	0.011

[1]n: number of individuals analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Table 4.12.2 Summary statistics for copper concentrations in sparrowhawk liver (mg/kg dry weight)[1]

Year	n	Mean	SD	Median	Min	Max	Q1	Q3
2007	18	1.321	0.173	1.311	1.021	1.700	1.195	1.463
2008	26	1.271	0.253	1.191	0.978	1.953	1.057	1.436
2009	19	1.181	0.179	1.190	0.868	1.508	1.004	1.312
2010	26	1.277	0.192	1.237	1.033	1.688	1.133	1.432
2011	22	1.271	0.221	1.193	1.013	1.857	1.114	1.408
2012	22	1.267	0.173	1.256	0.969	1.606	1.121	1.435

Year	n	Mean	SD	Median	Min	Max	Q1	Q3
2013	22	1.214	0.171	1.154	1.013	1.616	1.082	1.303
2014	20	1.334	0.245	1.204	1.086	1.734	1.112	1.600

[1]n: number of individuals analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Table 4.12.3 Summary statistics for zinc concentrations in sparrowhawk liver (mg/kg dry weight)[1]

Year	n	Mean	SD	Median	Min	Max	Q1	Q3
2007	18	156	94.0	2.096	46.2	376	90.7	203
2008	26	142	93.2	1.997	55.8	432	62.0	205
2009	19	96.6	47.4	1.927	31.9	202	62.8	123
2010	26	134	65.4	2.053	61.5	271	75.2	174
2011	22	135	87.9	2.031	59.5	407	72.1	195
2012	22	155	96.1	2.047	71.1	378	85.3	212
2013	22	108	67.0	1.863	49.6	279	57.1	136
2014	20	94.5	44.8	1.821	52.6	198	57.6	135

[1]n: number of individuals analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

There was a negative correlation between nickel concentration and year (Spearman's rank correlation coefficient (r_s) = –0.153, p = 0.043) for 2007 to 2014, but this relationship was not linear and so the temporal trends could not be tested by a linear regression analysis. A dashboard entry of 'Decreasing concentrations' is recommended for nickel based on the correlation analysis. There was no short-term temporal trend (r_s = –0.156, p = 0.074) for years 2009 to 2014.

For copper there is no statistically significant relationship between concentration and year for both the full dataset and the short-term temporal analysis (r_s = –0.016 and 0.064, respectively, p >0.468) and no statistically significant difference among years (KW = 7.436, p = 0.384). For the dashboard, an entry of 'No observed change in concentrations' is used for copper.

Concentrations of zinc in liver were negatively correlated with year (r_s = –0.159, p = 0.036) for 2007 to 2014, but this relationship was not linear. For the dashboard, an entry of 'Decreasing concentrations' is recommended for zinc. There was no statistically significant short-term temporal trend (r_s = –0.119, p = 0.176) for the years 2009 to 2014.

As these metals are essential metals in avian species and they will be regulated in the body, the significance of these trends is not yet clear.

4.12.4 Thresholds

No threshold values for residues of nickel, copper and zinc in birds' liver are available. Therefore, the entry in the dashboard reflects that there is no value available for comparison.

4.13 Heavy metals in freshwater: lead and cadmium

Lead
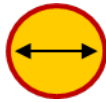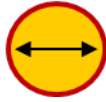
Cadmium

4.13.1 Data source

Data on bioavailable lead and dissolved cadmium concentrations in water have been provided by the Environment Agency from their freshwater statutory monitoring network.

Bioavailable lead is a calculated value based on dissolved lead concentrations and dissolved organic carbon (DOC) content at a site (wca-environment, 2014; UKTAG, 2014). Where the DOC value has not been derived from the same sample as the analysed dissolved metal concentration, site means are used for the bioavailability calculation. Most results have an analysed DOC value from 2014. The bioavailable lead concentrations are calculated using the lead screening tool (UKTAG, 2014).

4.13.2 Data structure

Relevant data are available for the period 2014–2019 for lead and cadmium in freshwaters across England. The data vary both in terms of the number of measurements taken within a year per site and the number of sites monitored per year. Some sites have been sampled in multiple years.

A data summary is available for each year based on the total number of measurements made in a year – that is all data pooled from all sites (see Tables 4.13.1 and 4.13.2). Summaries are also available for each site based on samples taken over the most recent 3 years and for which there were more than 3 samples per year.

Bioavailable lead is a calculated value and so there is no corresponding LoD. The dissolved lead results which were used in that calculation that were below the LoD were taken at half their face value. The LoD for dissolved lead is 0.1µg/L. The LoDs for dissolved cadmium are variable and predominantly range from 0.01 to 0.1µg/L with a nominal amount of samples reported up to a value of 5µg/L. Results recorded as below the LoD were assigned a value equal to half the LoD.

Analytical methods with lower LoDs have been introduced over time, in particular from 2014, and monitoring sites have been continuously reviewed – removing sites that do not show contamination issues and including new ones where a potential source of contamination has been newly identified. For these reasons we have selected the period 2014–2019 for reporting here to maintain the integrity of the time series, although data are available prior to that. It should also be noted that while analytical methods have been

improved, older methods may have been used for a limited number of more-recent datapoints for cadmium.

4.13.3 Exploration of change in chemical concentrations over time

A summary of how bioavailable lead and dissolved cadmium concentrations in freshwater in England have varied over time is presented in Tables 4.13.1 and 4.13.2 and Figures 4.13.1 and 4.13.2, respectively.

The presentation of data in the figures differs to that for some of the other data sources in this report because the large number of samples or even sites for which data are available would otherwise result in a cluttered figure. For the purposes of clarity, the data in Figures 4.13.1 and 4.13.2 are presented as annual median, interquartile range and 10–90th percentiles of individual sample concentrations.

Table 4.13.1 Summary statistics for concentrations of bioavailable lead in samples at all freshwater monitoring sites (µg/L)[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2014	1147	9966	0.290	1.00	0.0893	0.00289	28.0	0.0313	0.229
2015	1291	10134	0.263	1.04	0.0497	0.00250	33.5	0.0234	0.160
2016	1222	8187	0.319	1.45	0.0446	0.00250	30.7	0.0224	0.113
2017	920	6560	0.344	1.44	0.0468	0.00250	28.3	0.0242	0.123
2018	763	5675	0.352	1.56	0.0434	0.00250	40.7	0.0225	0.122
2019	759	5136	0.302	1.29	0.0455	0.00263	34.3	0.0226	0.131

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Table 4.13.2 Summary statistics for concentrations of dissolved cadmium in samples at all freshwater monitoring sites (µg/L)[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2014	1151	10014	0.0905	0.341	0.0464	0.00500	8.60	0.0125	0.0500

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2015	1307	10025	0.0847	0.280	0.0248	0.00500	6.85	0.0105	0.0500
2016	1240	8331	0.108	0.438	0.0241	0.00500	19.0	0.0105	0.0547
2017	937	6814	0.123	0.518	0.0233	0.00500	20.5	0.0105	0.0611
2018	781	5869	0.128	0.395	0.0246	0.00500	7.51	0.0105	0.0662
2019	778	5321	0.107	0.406	0.023	0.00500	16.0	0.00500	0.0540

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

The national datasets for bioavailable lead and dissolved cadmium contain a subset of data for rivers which are polluted by abandoned metal mines. Cleaning up pollution from such sites – 'WAMM' sites identified by the Water and Abandoned Metal Mines Programme – has been highlighted as beneficial to the environment in the 25-YEP. For this reason, we have included plots of data for samples taken from WAMM sites and for those from non-WAMM ones alongside our national overview of all sites in Figures 4.13.1 and 4.13.2. The locations of the WAMM sites are shown in Appendix C.

Simple visual inspection of the data for the WAMM versus the non-WAMM sites indicates that they clearly differ (Figures 4.13.1 and 4.13.2). Average values (mean and median) are over an order of magnitude higher at the WAMM sites with known metals' contamination than at non-WAMM ones. The median concentration levels appear fairly similar throughout the years examined.

For the purposes of providing national trend assessment data for the dashboard, all sample data were considered in the temporal trend analysis. However, additional assessments were performed based on sample data from WAMM and non-WAMM sites to determine if there were any statistically significant variations over time for these data subsets.

The geometric mean was taken of all samples at all sites per year for each metal to minimise any skews in the data and the undue influence of outliers. Seasonal patterns were considered to minimise the impact of quality varying throughout the year owing to natural cycles; measurements were grouped into four seasons: winter (January to March), spring (April to June), summer (July to September), and autumn (October to December). The tseries package in R was used to analyse the time series and identify any potential trends, the significance of which was assessed using the Cox Stuart trend test. Statistically significant trends were those for which the p-value was <0.05.

Figure 4.13.1 Median, interquartile range and 10–90th percentiles of bioavailable lead concentrations in freshwater (µg/L) for samples taken from all sites, WAMM sites and non-WAMM sites
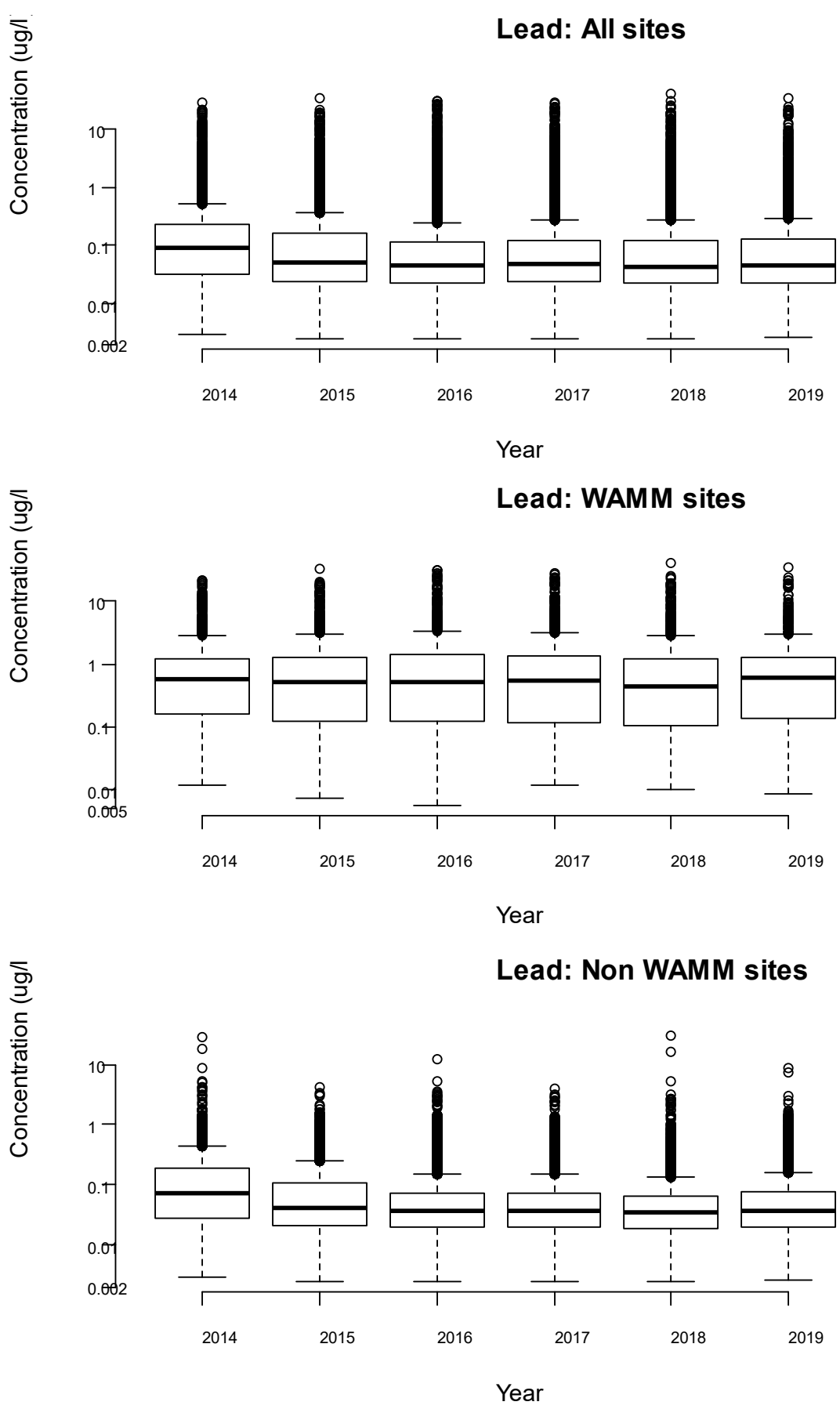
Figure 4.13.2 Median, interquartile range and 10–90th percentiles of dissolved cadmium concentrations in freshwater (μg/L) for samples taken from all sites, WAMM sites and non-WAMM sites
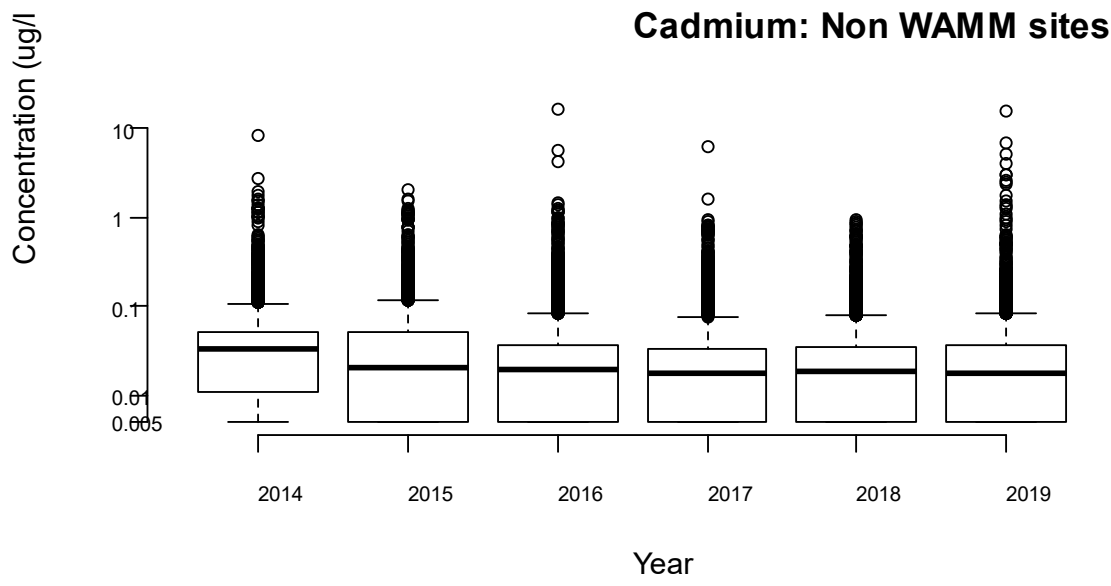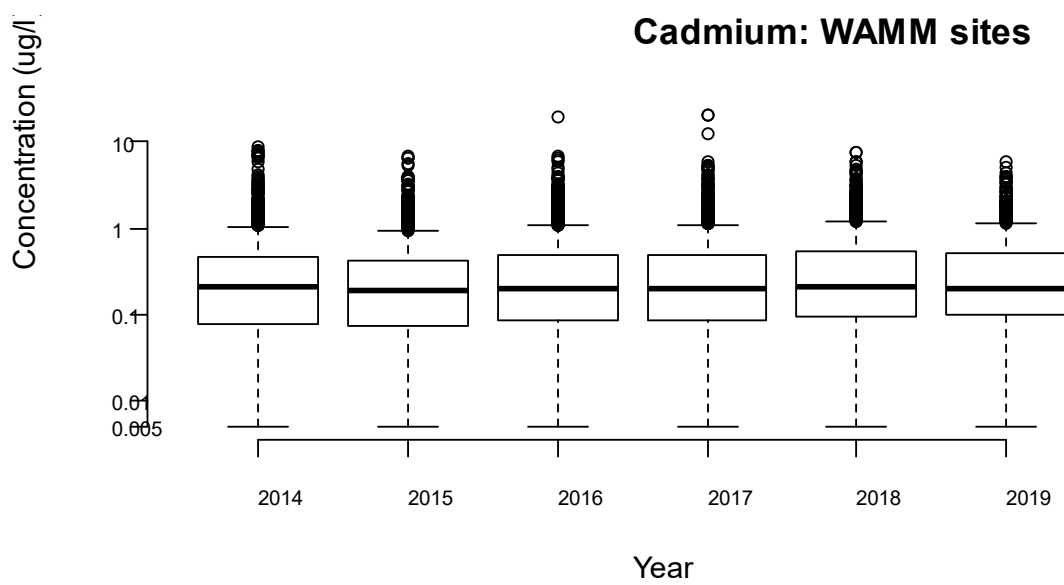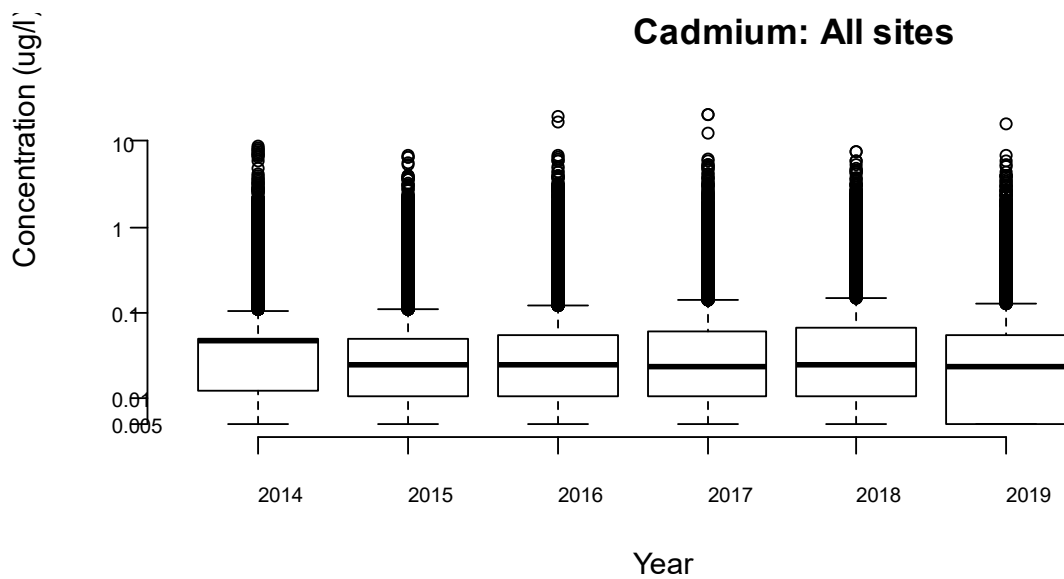
Assessments based on medians and geomeans of sites and were also performed but there was very little difference in the results.

Figures 4.13.3 and 4.13.4 show the breakdown of the time series data for lead and cadmium, respectively, based on seasonal geometric means of all samples from all sites, WAMM sites and non-WAMM sites. For each of these three cases, there are four graphs illustrating the raw data as a seasonal value ('data'), the effects of the seasonal pattern (repeating the four seasons for each year; 'seasonal'), the trend ('trend'), and the residual concentrations once the seasonal and trend series data are removed ('remainder').

Assessment of the data shows that there are no statistically significant trends for any of the three datasets based on the geometric means for bioavailable lead (Table 4.13.3). For dissolved cadmium, the situation is similar for all and WAMM sites. However, non-WAMM sites exhibit a downward trend ($p = 0.019$) (Table 4.13.3).

Table 4.13.3 Summary of p-values from the temporal trend assessment of the geometric means of bioavailable lead and dissolved cadmium concentrations in freshwaters

| Substance	Type of sites assessed	p-value any trend	p-value downward trend	p-value upward trend	Decision
Lead	all	0.39	0.19	0.93	no change
Lead	WAMM	0.15	0.073	0.98	no change
Lead	non-WAMM	0.77	0.39	0.81	no change
Cadmium	all	0.74	0.81	0.39	no change
Cadmium	WAMM	0.15	0.98	0.073	no change
Cadmium	non-WAMM	0.039	0.019	1.0	downward trend

The dashboard trend information is based on the overall national assessment; therefore, the corresponding entry is 'No observed change in concentrations', but the result relating to the non-WAMM sites for cadmium is worth noting.

Figure 4.13.3 Breakdown of trend analysis of seasonal geometric means of bioavailable lead (µg/L) in freshwater
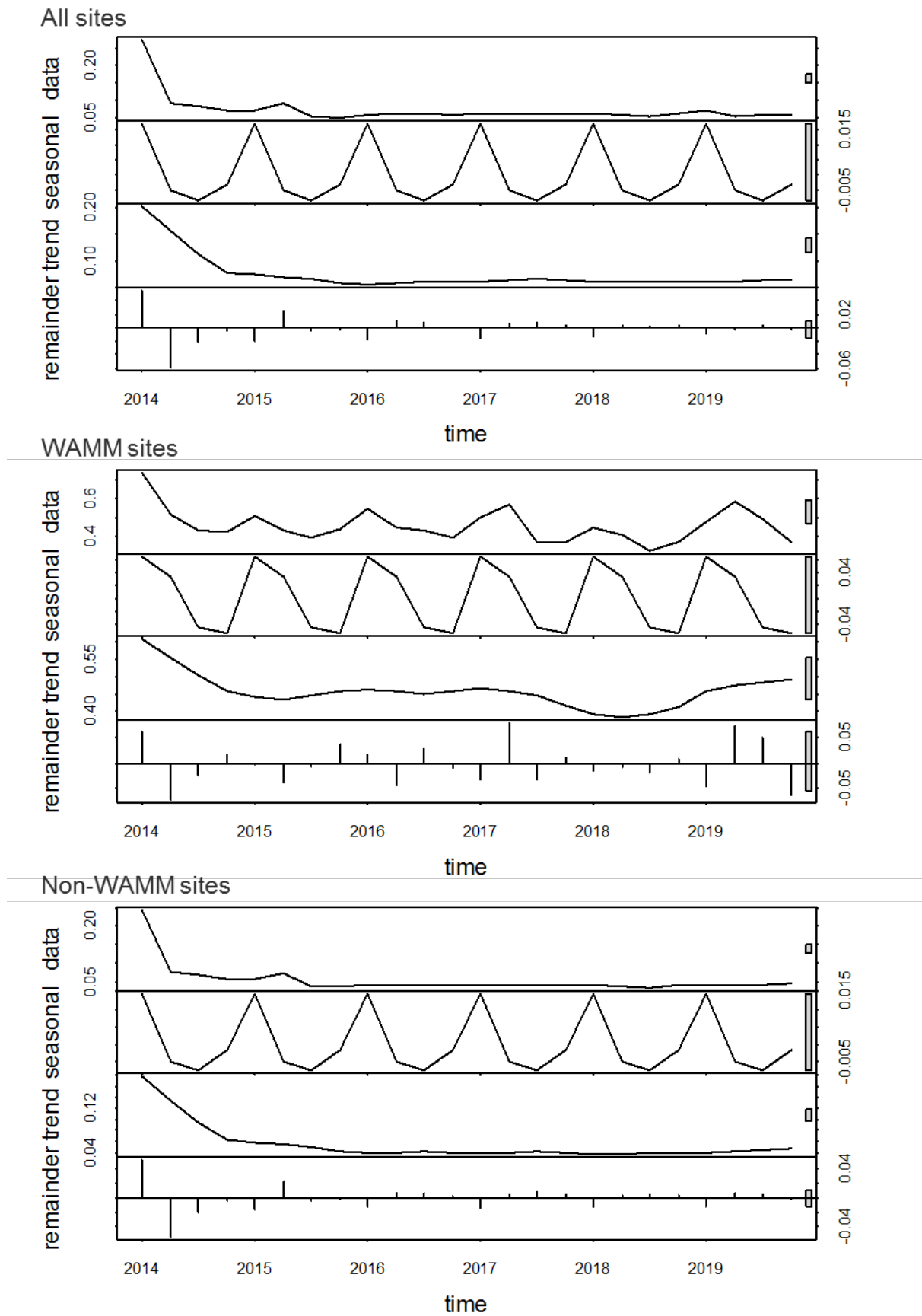
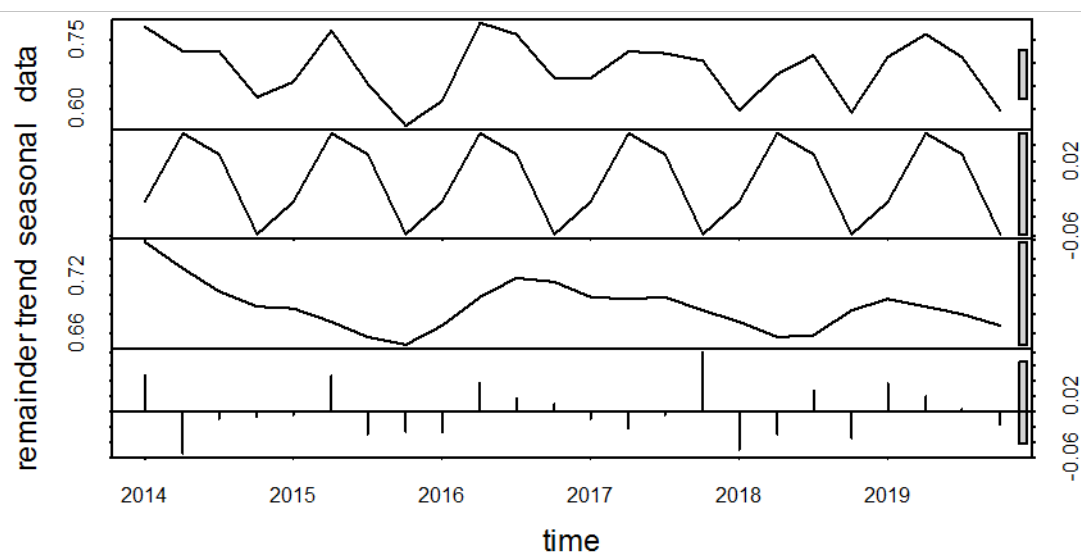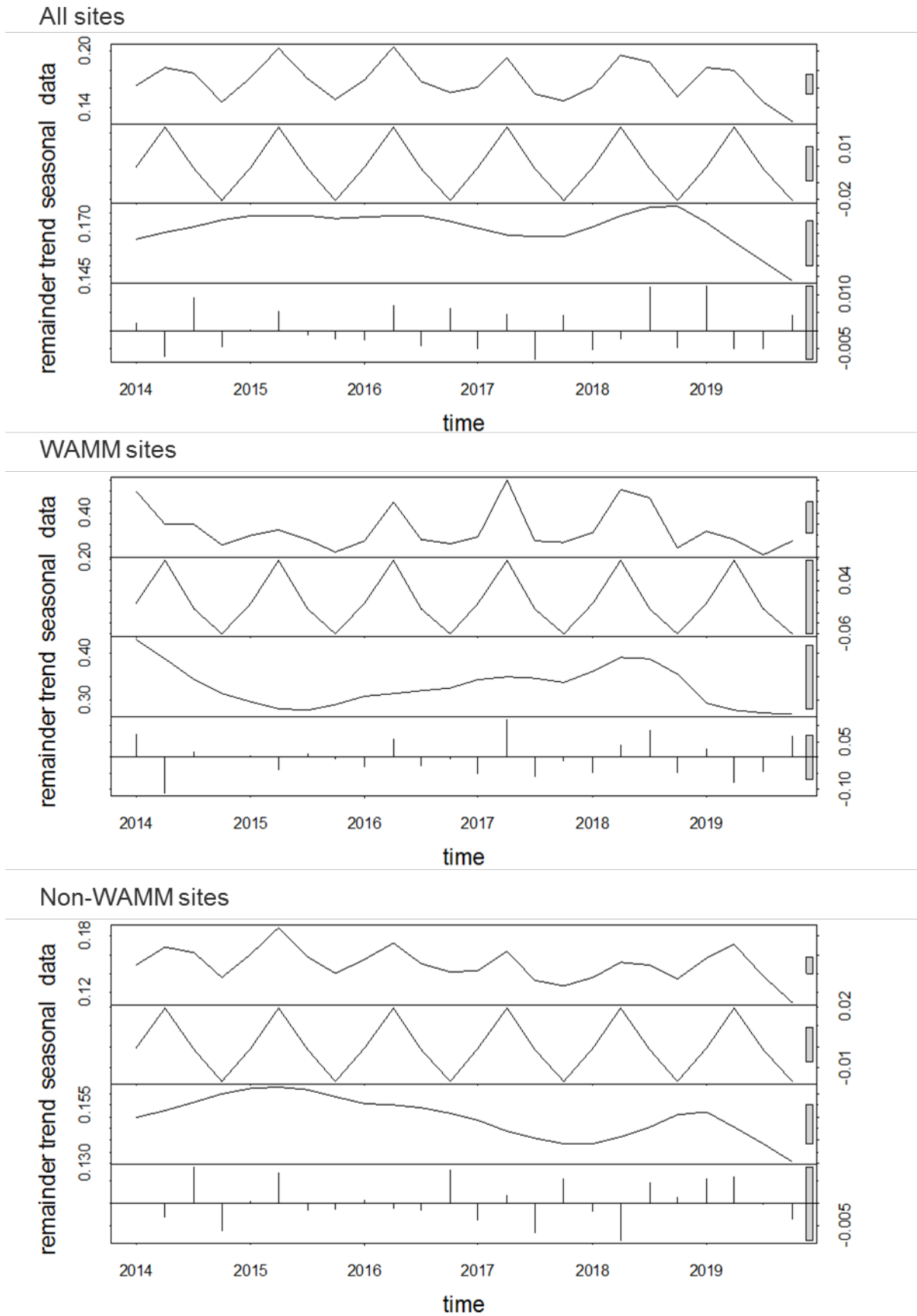**Figure 4.13.4 Breakdown of trend analysis of seasonal geometric means of dissolved cadmium (µg/L) in freshwater**
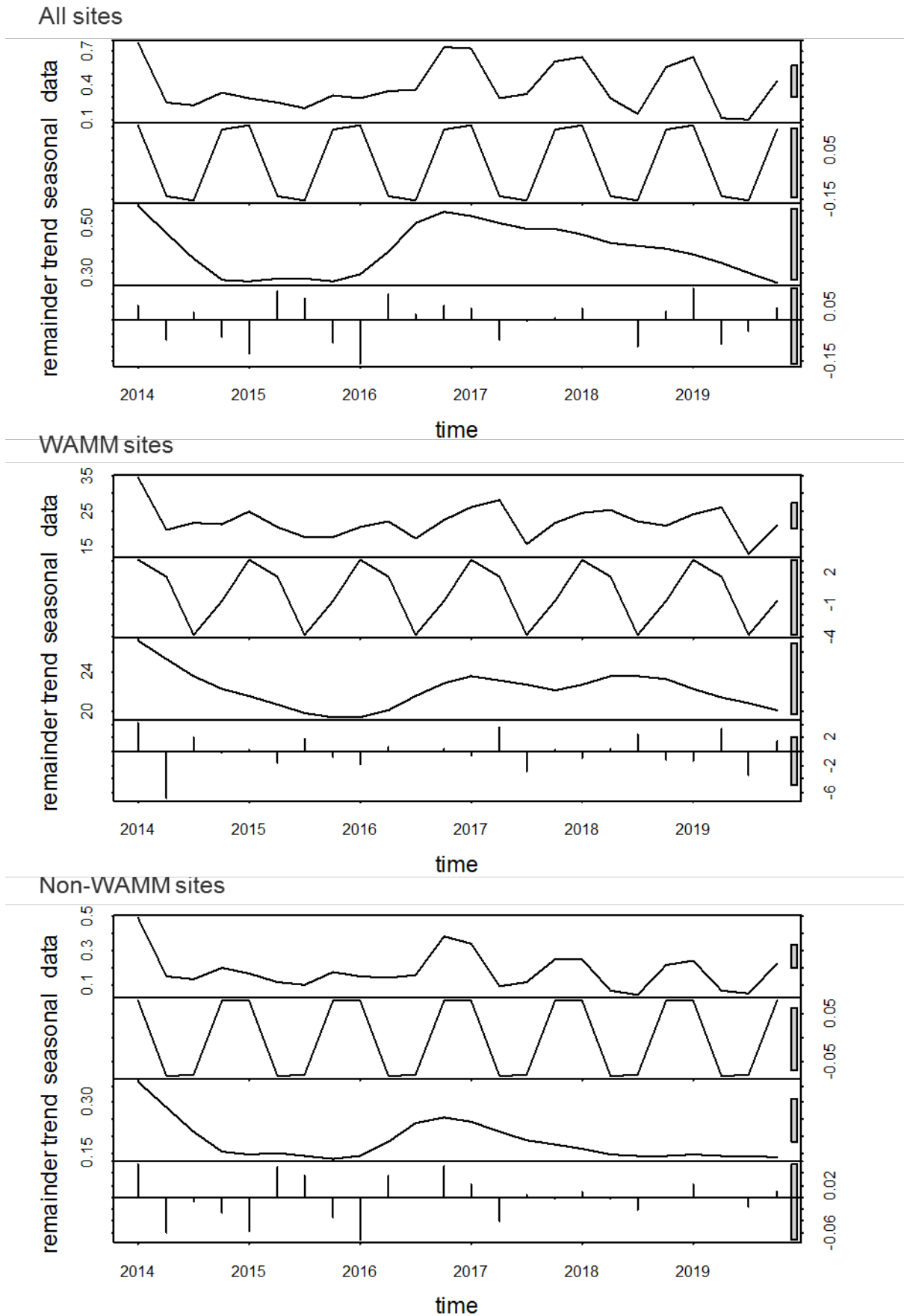

All sites
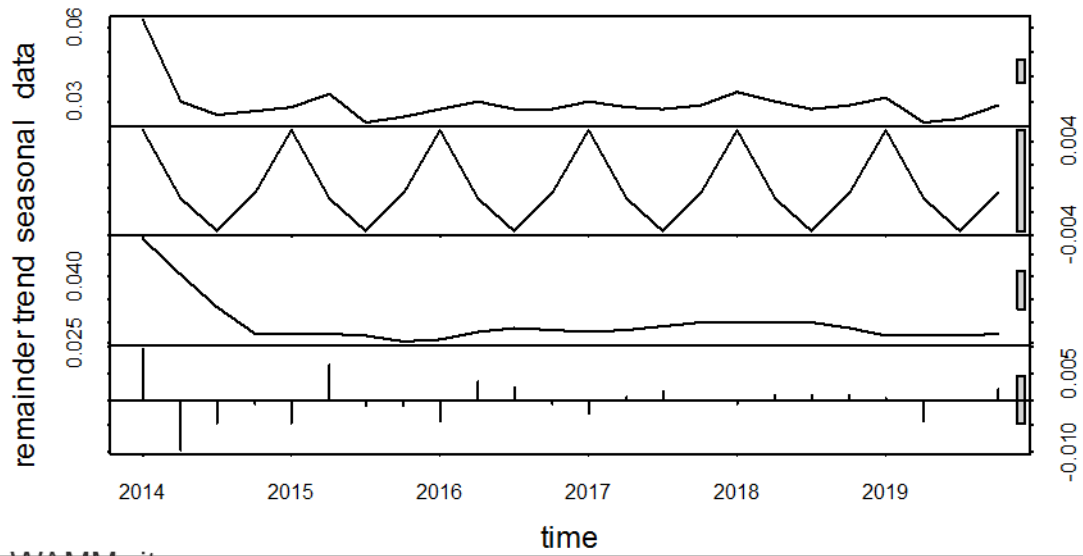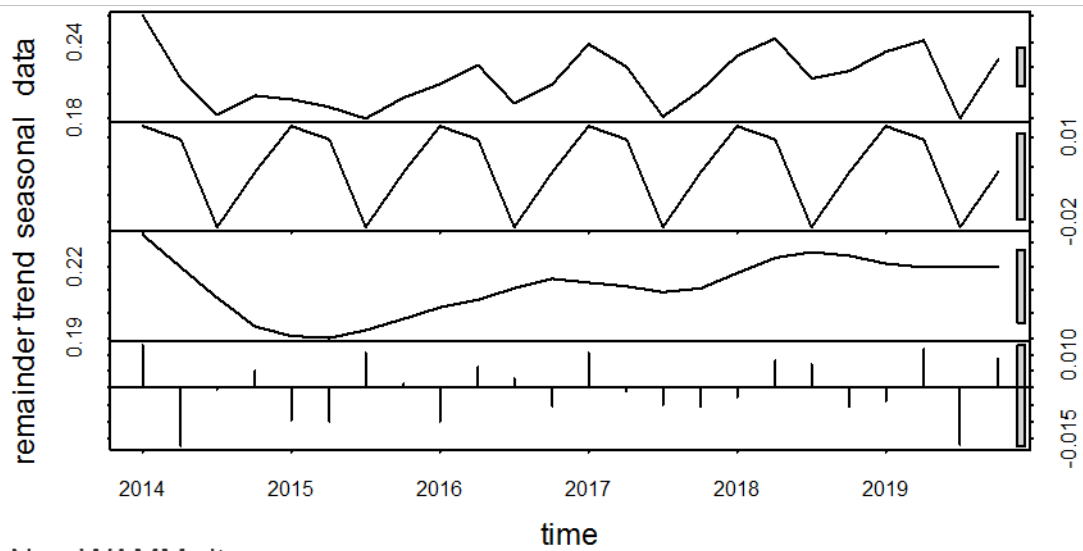
WAMM sites

Non-WAMM sites
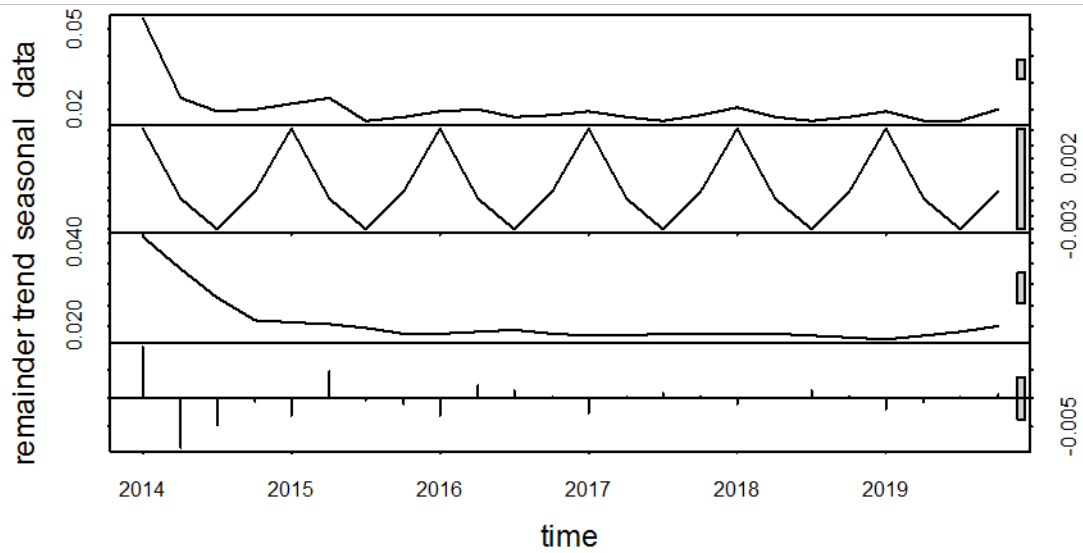
4.13.4 Thresholds

An annual average EQS value of 1.2μg/L for bioavailable lead in inland surface waters is given in the Water Framework Directive (Standards and Classification) Directions 2015 (UK Government, 2015). For dissolved cadmium, the EQS values vary between ≤0.08 and 0.25μg/L depending on the hardness of the water (UK Government, 2015). These thresholds are given in Table 4.13.4 and are used for the assessment here.

Table 4.13.4 Annual average environmental quality standards for bioavailable lead and dissolved cadmium in freshwater (UK Government, 2015) and the proportion of sites above these values for 2017–2019[1]

Substance	Hardness (mg $CaCO_3$/L)	EQS (μg/L)	Number of sites above EQS	Proportion of sites above EQS (%)
Bioavailable lead	–	1.2	32	3.8
Dissolved cadmium	0 to <50	0.08	13	1.5
Dissolved cadmium	50 to <100	0.009	36	4.2
Dissolved cadmium	100 to <200	0.15	21	2.4
Dissolved cadmium	≥200	0.25	22	2.6

[1] $CaCO_3$: calcium carbonate; EQS: environmental quality standard.

Typically, average site concentrations are used for comparison with the EQSs for lead and cadmium. These are based on available data for a 3-year period. The assessment here is based on site averages for the period 2017–2019. Not every site has the maximum number of years' data available. Each site requires >3 samples taken over that period to be included in the assessment; the number of samples per site varied between 4 and 44 for both lead and cadmium.

The number and proportion of sites with mean concentrations for 2017–2019 that exceed the above thresholds have been calculated. For lead, 32 out of 842 sites (3.8%) had mean concentrations above the threshold of 1.2μg/L. For cadmium, a slightly higher rate of exceedance was seen: 92 out of 861 sites (11%) were above the relevant thresholds given in Table 4.13.3. The percentage results are used for the corresponding entries in the dashboard.

It is noteworthy that over four-fifths of the results above the EQSs were observed at WAMM sites for lead (28 sites) and cadmium (75 sites) indicating them to be key areas for improvement.

4.14 Heavy metals in freshwater: nickel, copper and zinc

Nickel

Copper

Zinc

4.14.1 Data source

Data on bioavailable nickel, copper and zinc have been provided by the Environment Agency derived from analysis of samples from their freshwater statutory monitoring network.

Bioavailable metals concentrations are calculated values based on the corresponding dissolved metal concentrations and pH, calcium and dissolved organic carbon (DOC) content at a site (UKTAG, 2014). Where those physico-chemical parameters have not been derived from the same sample as the analysed dissolved metal concentration, site means are used for the bioavailability calculation. Most results have analysed values from 2014. The bioavailable metal concentrations are calculated using the Metals Bioavailability Assessment Tool (M-BAT) (UKTAG, 2014).

In the case of zinc, ambient background concentrations (ABCs) have been removed from dissolved zinc measurements before using the M-BAT to allow for any species acclimatisation to such levels. The corresponding threshold for zinc allows for this 'added risk' approach; that is, it is a threshold relating to the concentration over and above the ABC (see Section 4.14.4) (UKTAG, 2014).

4.14.2 Data structure

Relevant data are available for the period 2014–2019 for bioavailable nickel, copper and zinc in freshwaters across England. The data vary both in terms of the number of measurements taken within a year per site and the number of sites monitored per year. Some sites have been sampled in multiple years.

A data summary is available for each year based on the total number of measurements made in a year – that is all data pooled from all sites (see Tables 4.14.1, 4.14.2 and 4.14.3). Summaries are also available for each site based on samples taken over the most recent 3 years and for which there were more than 3 samples per year.

The bioavailable metal concentrations are calculated values and so have no corresponding LODs, but dissolved metal results which were used in those calculations that were below the LoD were taken at half their face value. The current LoDs for dissolved nickel, copper and zinc are 0.5, 1 and 0.5µg/L, respectively.

Analytical methods with lower LoDs have been introduced over time for nickel and zinc, in particular from 2014, and monitoring sites have been continuously reviewed – removing sites that do not show contamination issues and including new ones where a potential source of contamination has been newly identified. For these reasons we have selected the period 2014–2019 for reporting here to maintain the integrity of the time series, although data are available prior to that.

4.14.3 Exploration of change in chemical concentrations over time

A summary of how bioavailable nickel, copper and zinc concentrations in freshwater in England have varied over time is presented in Tables 4.14.1, 4.14.2 and 4.14.3 and Figures 4.14.1, 4.14.2 and 4.14.3, respectively.

The presentation of data in the figures differs to that for some of the other data sources in this report because the large number of samples or even sites for which data are available would otherwise result in a cluttered figure. For the purposes of clarity, the data in Figures 4.14.1, 4.14.2 and 4.14.3 are presented as annual median, interquartile range and 10–90th percentiles of individual sample concentrations.

Table 4.14.1 Summary statistics for concentrations of bioavailable nickel in freshwaters (µg/L)[1]

| Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2014	1220	10392	1.24	2.80	0.691	0.0406	104	0.446	1.10
2015	1307	10266	1.25	2.60	0.674	0.0293	59.2	0.387	1.14
2016	1242	8376	1.30	3.72	0.713	0.0300	276	0.421	1.23
2017	942	6708	1.41	3.45	0.727	0.0275	173	0.418	1.25
2018	787	5888	1.33	2.55	0.693	0.0397	55.8	0.391	1.25
2019	785	5340	1.18	2.70	0.722	0.0471	127	0.439	1.18

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Table 4.14.2 Summary statistics for concentrations of bioavailable copper in freshwaters (µg/L)[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2014	1304	10376	0.787	11.8	0.158	0.00150	543	0.103	0.250
2015	1386	10229	0.666	7.81	0.162	0.00311	337	0.105	0.262
2016	1307	8346	1.05	14.0	0.155	0.00115	482	0.101	0.259
2017	1011	6609	0.761	7.00	0.144	0.00331	244	0.0903	0.247
2018	862	5752	1.90	20.7	0.148	0.00373	415	0.0962	0.253
2019	865	5224	1.06	14.1	0.140	0.00399	444	0.0894	0.239

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Table 4.14.3 Summary statistics for concentrations of bioavailable zinc in freshwaters (µg/L)[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2014	1120	9298	15.4	85.8	1.76	1.41×10^{-8}	2136	0.448	4.92
2015	1335	9570	15.1	76.1	1.92	1.13×10^{-8}	1347	0.469	5.71
2016	1266	8025	18.7	87.5	2.32	1.26×10^{-8}	2299	0.653	7.06
2017	961	6481	23.9	96.4	2.53	1.11×10^{-8}	1421	0.727	8.48
2018	811	5605	29.2	126	2.43	1.41×10^{-8}	1872	0.616	8.95
2019	809	5003	16.5	79.1	1.82	1.14×10^{-8}	1614	0.490	5.59

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

The national datasets for bioavailable nickel, copper and zinc contain a subset of data for rivers which are polluted by abandoned metal mines. Cleaning up pollution from such sites – 'WAMM' sites identified by the Water and Abandoned Metal Mines Programme – has been highlighted as beneficial to the environment in the 25-YEP. For this reason, we have included plots of data for samples taken from WAMM sites and for those from non-WAMM ones alongside our national overview of all sites in Figures 4.14.1, 4.14.2 and 4.14.3. The locations of the WAMM sites are shown in Appendix C.

There are no distinct differences in annual average (mean and median) bioavailable nickel values for WAMM versus non-WAMM sites; such concentrations for WAMM sites are only marginally higher (Figure 4.14.1). A greater range of concentration values is seen at non-WAMM sites. For bioavailable copper, the annual median values are also fairly similar for both types of site (Figure 4.14.2). However, the annual means at WAMM sites are an order of magnitude higher, driven by the greater extreme values observed. Values for bioavailable zinc at WAMM sites are generally well over an order of magnitude higher than those at non-WAMM sites (Figure 4.14.3).

For the purposes of providing national trend assessment data for the dashboard, all sample data were considered in the temporal trend analysis. However, additional assessments were performed based on sample data from WAMM and non-WAMM sites to determine if there were any statistically significant variations over time for these data subsets.

The geometric mean was taken of all samples at all sites per year for each metal to minimise any skews in the data and the undue influence of outliers. Seasonal patterns were considered to minimise the impact of quality varying throughout the year owing to natural cycles; measurements were grouped into four seasons: winter (January to March), spring (April to June), summer (July to September), and autumn (October to December). The tseries package in R was used to analyse the time series and identify any potential trends, the significance of which was assessed using the Cox Stuart trend test. Statistically significant trends were those for which the p-value was <0.05.

Assessments based on medians and geomeans of sites and were also performed but there was very little difference in the results.

Figure 4.14.1 Median, interquartile range and 10–90th percentiles of bioavailable nickel concentrations in freshwater (µg/L) for samples taken from all sites, WAMM sites and non-WAMM sites
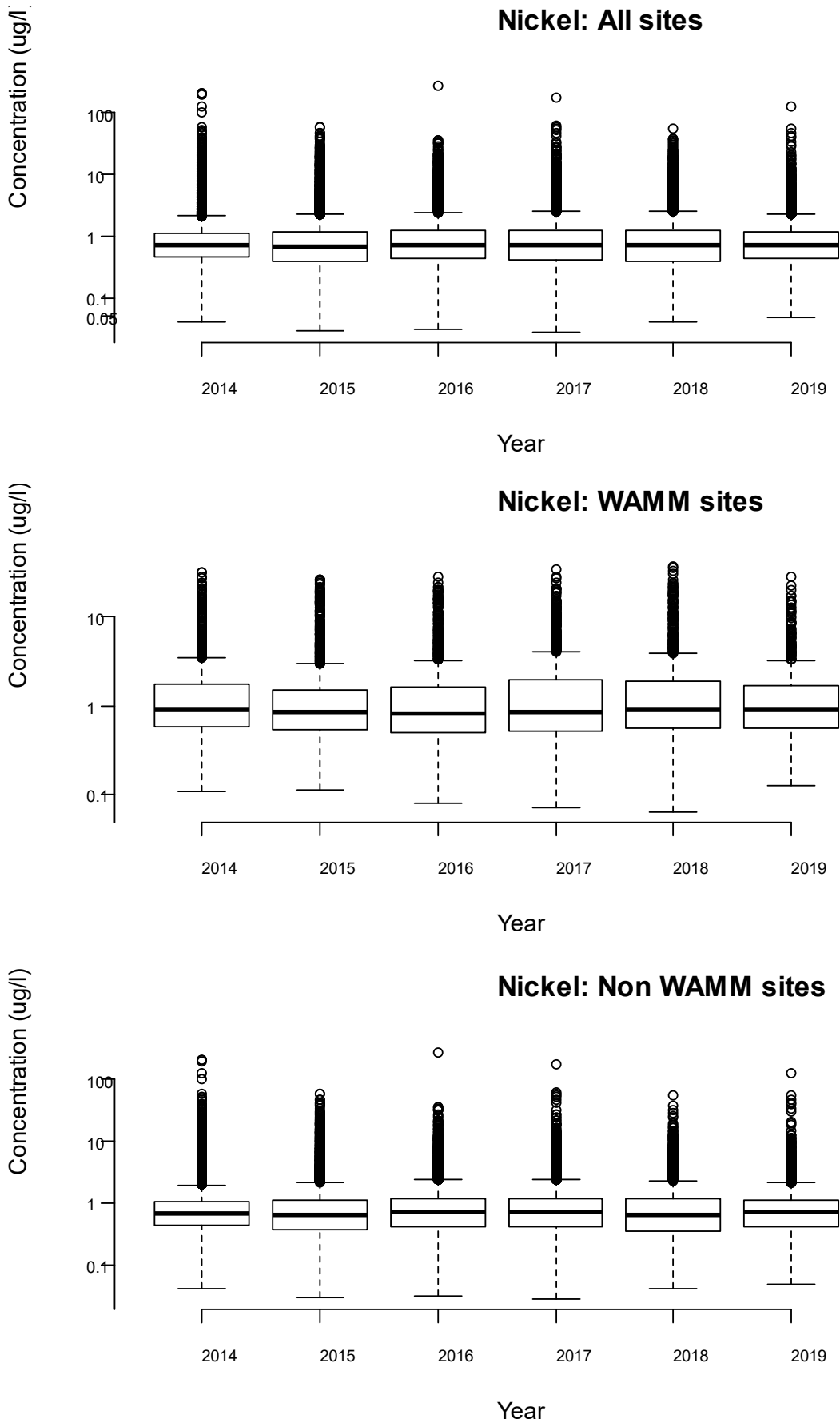
Figure 4.14.2 Median, interquartile range and 10–90th percentiles of bioavailable copper concentrations in freshwater (µg/L) for samples taken from all sites, WAMM sites and non-WAMM sites
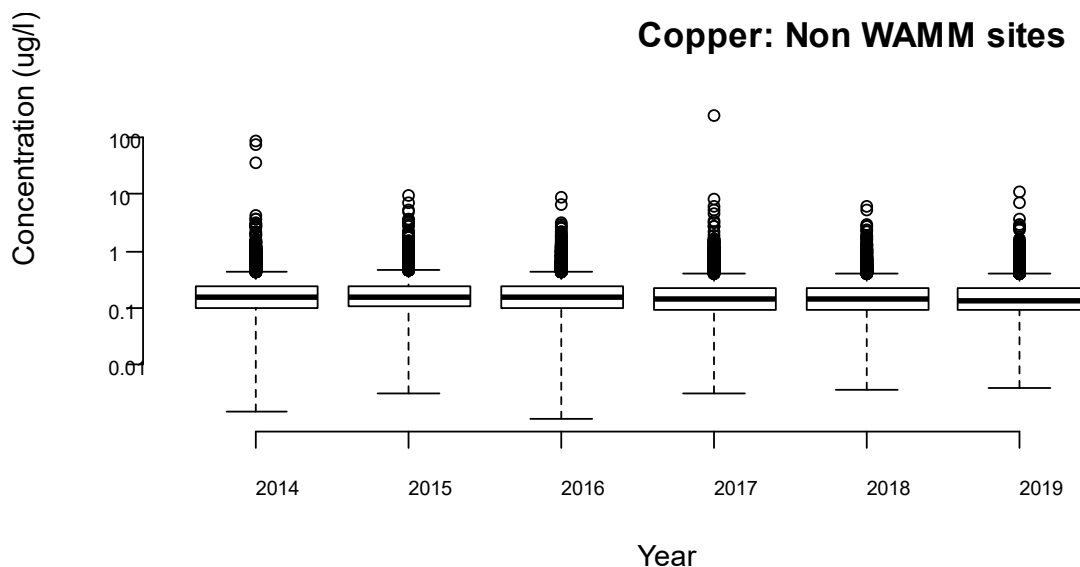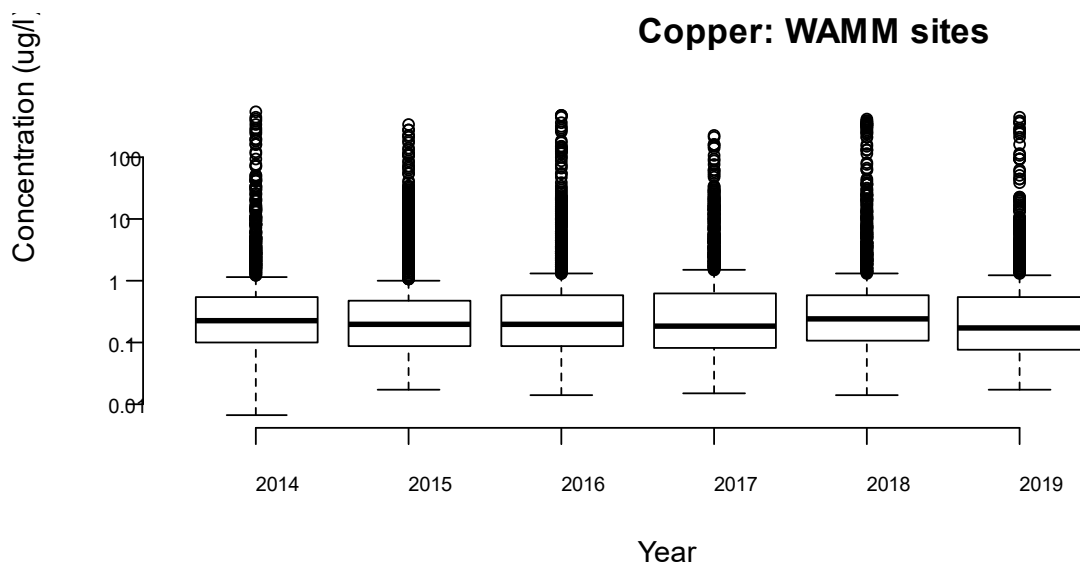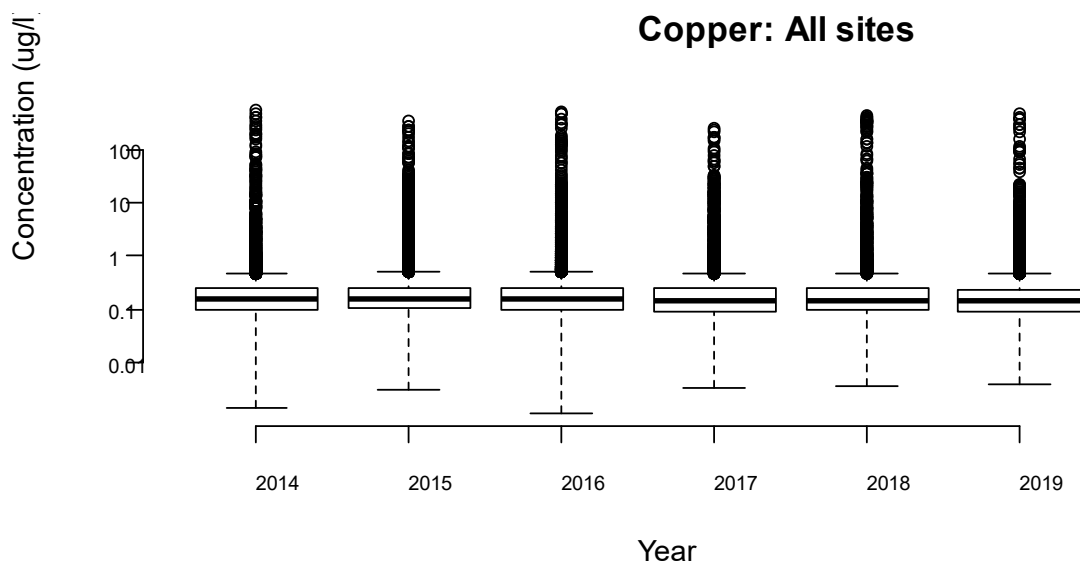
Figure 4.14.3 Median, interquartile range and 10–90th percentiles of bioavailable zinc concentrations in freshwater (µg/L) for samples taken from all sites, WAMM sites and non-WAMM sites
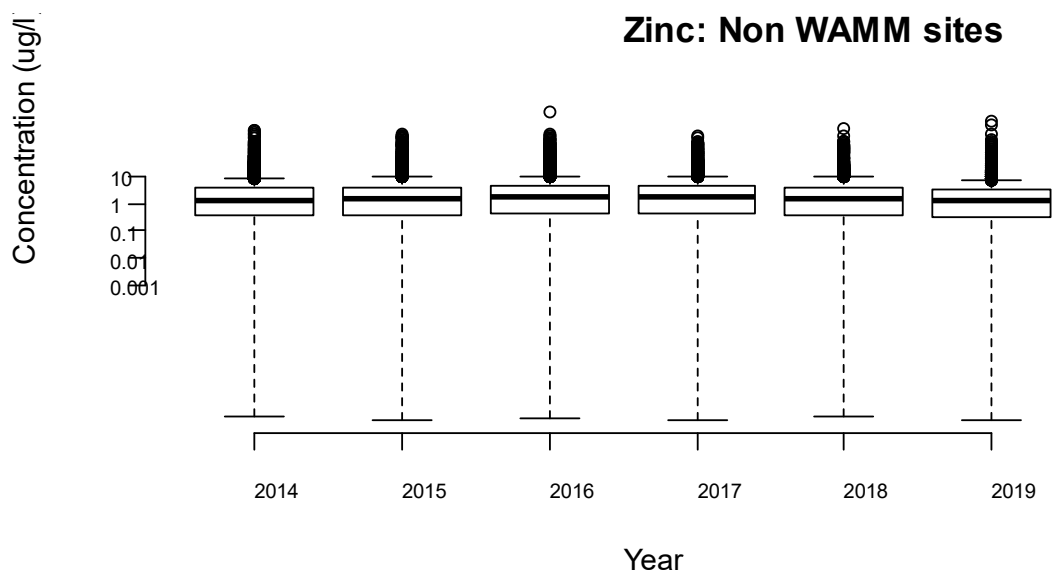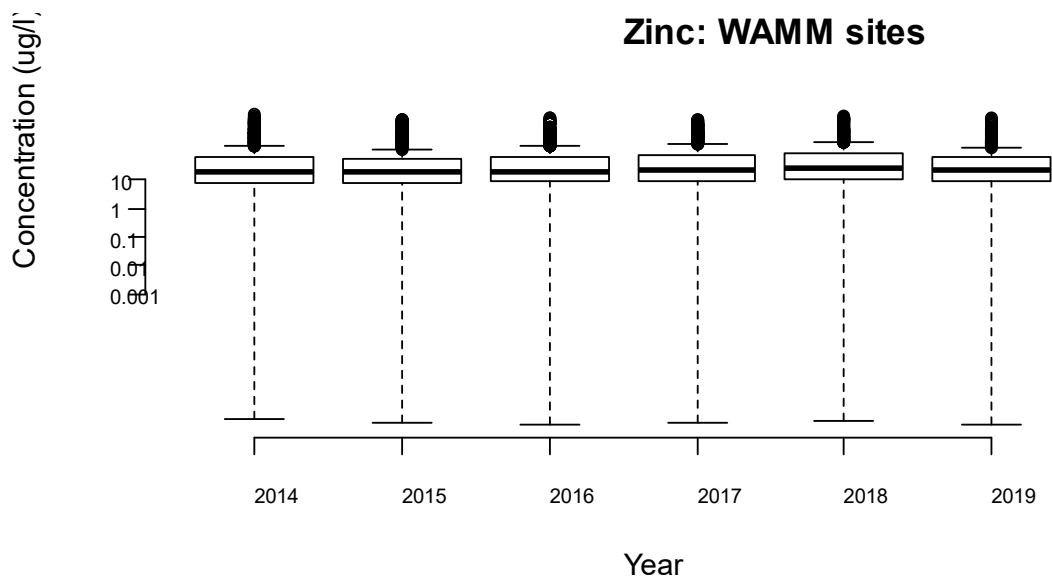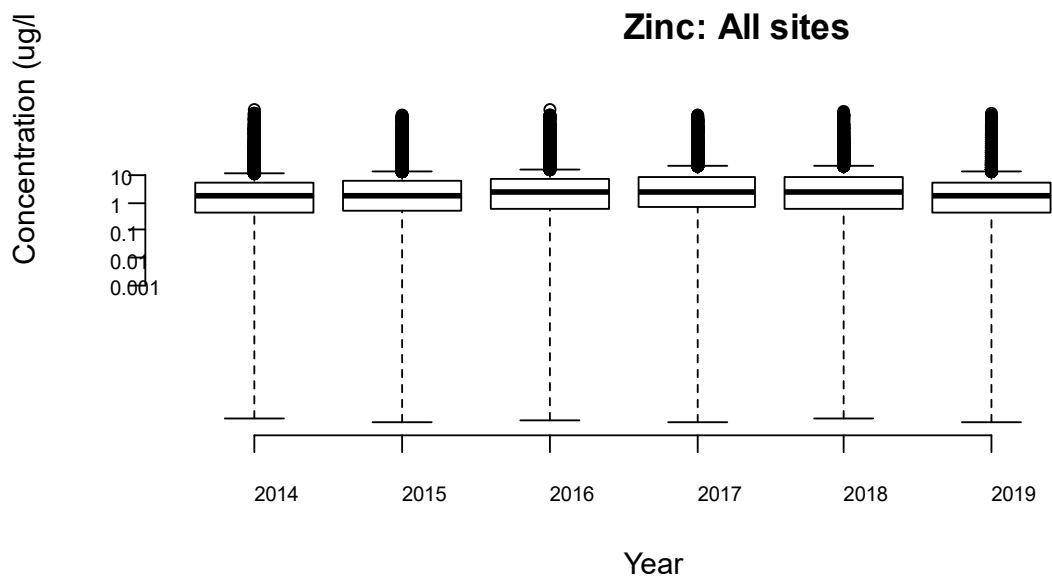
Figures 4.14.4, 4.14.5 and 4.14.6 show the breakdown of the time series data for nickel, copper and zinc, respectively, based on seasonal geometric means of all samples from all sites, WAMM sites and non-WAMM sites. For each of these three cases, there are four graphs illustrating the raw data as a seasonal value ('data'), the effects of the seasonal pattern (repeating the four seasons for each year; 'seasonal'), the trend ('trend'), and the residual concentrations once the seasonal and trend series data are removed ('remainder').

Assessment of the data shows that there are no statistically significant trends for any of the three datasets based on the geometric means for bioavailable nickel and zinc (Table 4.14.4). For bioavailable copper, the situation is similar for all and WAMM sites. However, non-WAMM sites exhibit a downward trend (p = 0.019) (Table 4.14.4).

Table 4.14.4 Summary of p-values from the temporal trend assessment of the geometric means of bioavailable lead and dissolved cadmium concentrations in freshwaters

Substance	Types of sites assessed	p-value any trend	p-value downward trend	p-value upward trend	Decision
Nickel	all	1.0	0.61	0.61	no change
Nickel	WAMM	0.39	0.93	0.19	no change
Nickel	non-WAMM	0.39	0.19	0.93	no change
Copper	all	0.77	0.39	0.81	no change
Copper	WAMM	0.39	0.93	0.19	no change
Copper	non-WAMM	0.0063	0.0032	1.0	downward trend
Zinc	all	0.77	0.81	0.39	no change
Zinc	WAMM	0.77	0.81	0.39	no change
Zinc	non-WAMM	0.38	0.19	0.93	no change

The dashboard trend information is based on the overall national assessment; therefore, the corresponding entries for nickel, copper and zinc are 'No observed change in concentrations', but the result relating to the non-WAMM sites for copper is worth noting.

Figure 4.14.4 Breakdown of trend analysis of seasonal geometric means of bioavailable nickel (μg/L) in freshwater

All sites
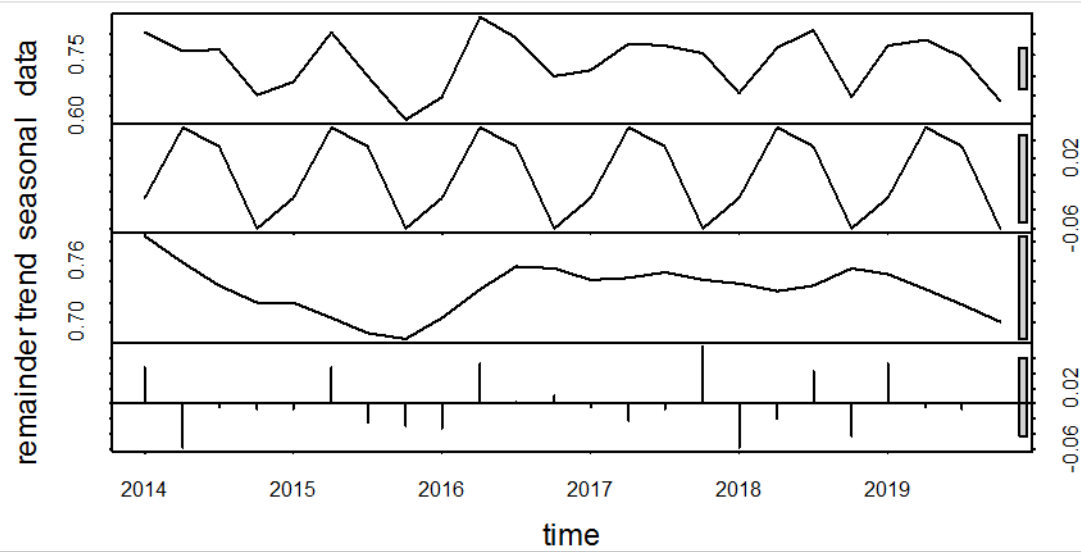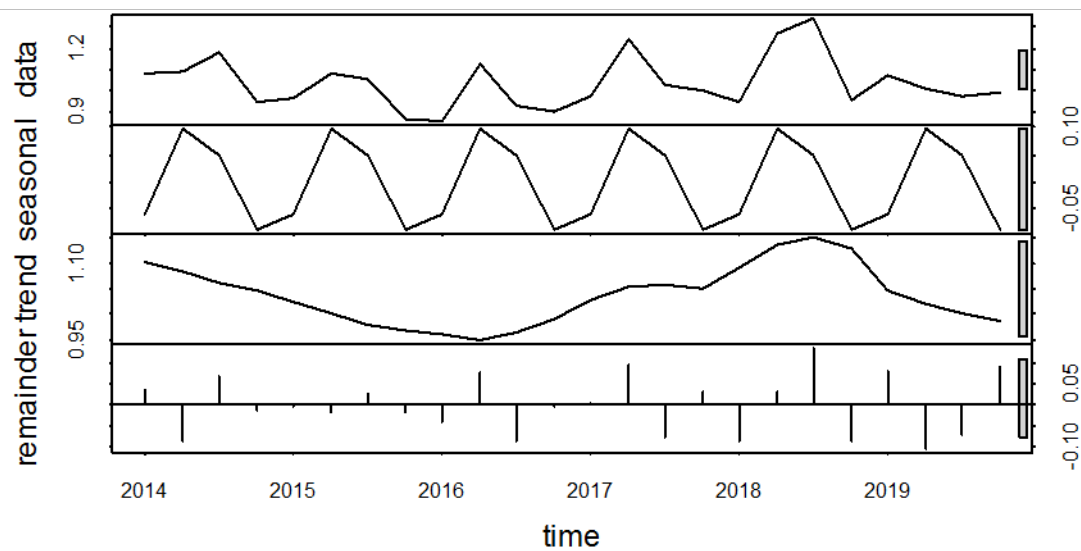
WAMM sites

Non-WAMM sites

Figure 4.14.5 Breakdown of trend analysis of seasonal geometric means of bioavailable copper (μg/L) in freshwater

Figure 4.14.6 Breakdown of trend analysis of seasonal geometric means of bioavailable zinc (µg/L) in freshwater

All sites

WAMM sites

Non-WAMM sites

4.14.4 Thresholds

An annual average EQS value of 4µg/L for bioavailable nickel in inland surface waters is given in the Water Framework Directive (Standards and Classification) Directions 2015 (UK Government, 2015). For copper and zinc, the EQSs are 1µg/L for bioavailable copper and 10.9µg/L for bioavailable zinc (plus any ABC of dissolved zinc; see Section 4.14.1) (UK Government, 2015). These EQSs are expressed as long-term means in freshwater.

Typically, average site concentrations are used for comparison with the EQSs for these metals. These are based on available data for a 3-year period. The assessment here is based on site averages for the period 2017–2019. Not every site has the maximum number of years' data available, although typically just under two-thirds of the sites had data for the 3 years. Each site requires >3 samples taken over that period to be included in the assessment; the number of samples per site varied between 4 and 44 for nickel and copper, and 4 and 45 for zinc.

The number and proportion of sites with mean concentrations for 2017–2019 that exceed the above thresholds have been calculated. For nickel, 39 out of 867 sites (4.5%) had mean concentrations above the threshold of 4µg/L. For copper, 26 out of 939 sites (2.8%) were above the EQS of 1µg/L for bioavailable copper. Zinc had the highest rate of exceedance with 145 out of 883 sites (16%) exceeding the threshold for the bioavailable fraction of the metal. These percentage results are used for the corresponding entries in the dashboard and reflect the results for all sites.

Approximately three-quarters of WAMM sites were above the EQS for zinc (91 sites out of 123), indicating that zinc concentrations present the highest risk at such sites compared with other metals (see also Section 4.13.4). However, there was still a good geographical spread of other higher risk sites: just under two-fifths of all sites show concentrations above the EQS at locations other than WAMM sites.

For nickel, risk was mainly attributed to other sites (27 sites) than WAMM ones. For copper, the opposite was true with 23 WAMM sites showing exceedances.

4.15 Heavy metals in freshwater fish: lead and cadmium

Lead   No dashboard entry – insufficient or no comparable data

Cadmium   No dashboard entry – insufficient or no comparable data

4.15.1 Data source

Data on lead and cadmium in fish in England have been provided by the Environment Agency. Concentration data in whole fish (roach, chub and brown trout) have been collected by the Environment Agency as part of its biota monitoring, which began in anticipation of requirements under the Water Environment Regulations 2017 (UK Government, 2017).

4.15.2 Data structure

Relevant data are available for the period 2016–2019 for lead and cadmium. The data consist of a variable number of measurements of the substances, both in terms of the number of freshwater fish sampled at a site and the number of sites monitored per year across England.

Individual sites are monitored on one occasion in the year. Typically, 5 fish are collected and analysed. However, the number of samples in the past have varied from 3 to 7 fish for both substances. The number of sites monitored in the earlier years is low. Some sites have been sampled in multiple years.

It should be noted that this data source is relatively new and a baseline dataset relating to designated trend sites is still being established. We have considered all site data as part of this assessment.

A data summary is available for each year based on the total number of measurements made in a year – that is all data pooled from all sites (see Table 4.15.1).

The LoDs given for lead vary with a value of 100μg/kg wet weight attributed to analysis of samples relating to 2016 and 2017. Following that, the LoDs were lower but based on dry weight samples analysed and converted into wet weight results. Around 20% of the results were reported as below the LoD. There were very few cases for cadmium where the result was below the LoD of approximately 2μg/kg wet weight. For lead and cadmium, results recorded as below the LoD were assigned a value equal to half the LoD.

4.15.3 Exploration of change in chemical concentrations over time

A summary of how lead and cadmium concentrations have varied over time is presented in Table 4.15.1 and in Figure 4.15.1.

Simple visual inspection of the data for all individuals may suggest that cadmium concentrations are beginning to show a gradual decline. However, there are too few data

to report a trend currently and assessment of what data were available showed no statistically significant change over time for either metals.

Table 4.15.1 Summary statistics for concentrations of lead and cadmium in whole freshwater fish (µg/kg wet weight)[1]

Substance	Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
Lead	2016	3	12	151	143	88.5	50.0	525	50.0	220
Lead	2017	4	19	156	225	69.9	47.2	1010	50.0	163
Lead	2018	11	52	266	405	105	11.7	2330	49.8	321
Lead	2019	16	79	152	301	72.6	10.7	2450	33.8	166
Cadmium	2016	3	12	18.7	7.73	15.9	9.36	35.0	14.0	22.7
Cadmium	2017	4	19	37.4	49.8	10.8	5.59	179	7.72	48.7
Cadmium	2018	11	52	13.8	15.5	6.92	1.86	61.0	4.01	16.9
Cadmium	2019	16	79	14.4	23.3	7.11	0.940	153	4.00	12.1

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1 lower interquartile range value; Q3: upper interquartile range value.

Figure 4.15.1 Median, interquartile range and 10–90th percentiles of lead and cadmium concentrations (μg/kg wet weight) in whole freshwater fish
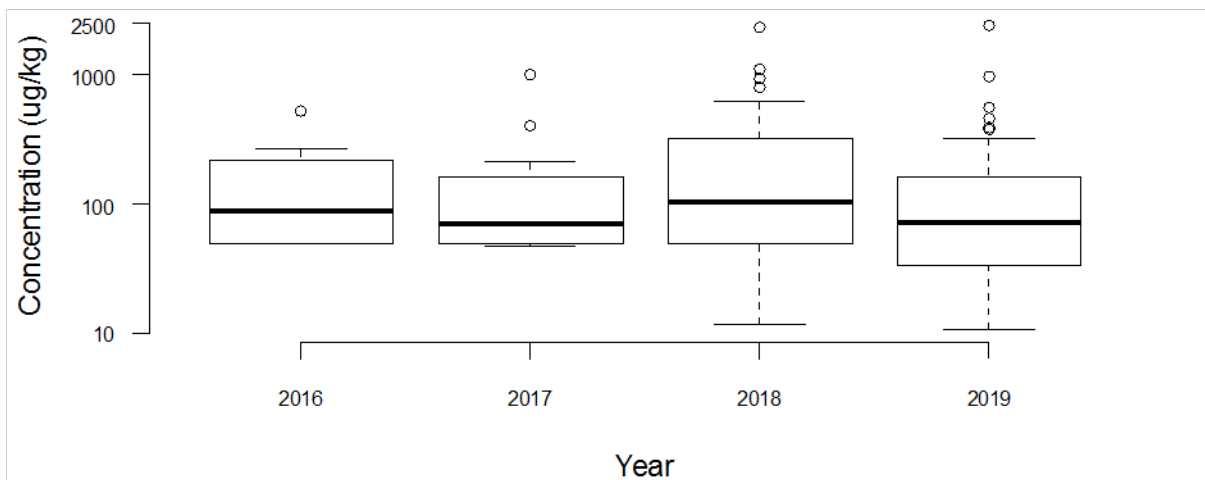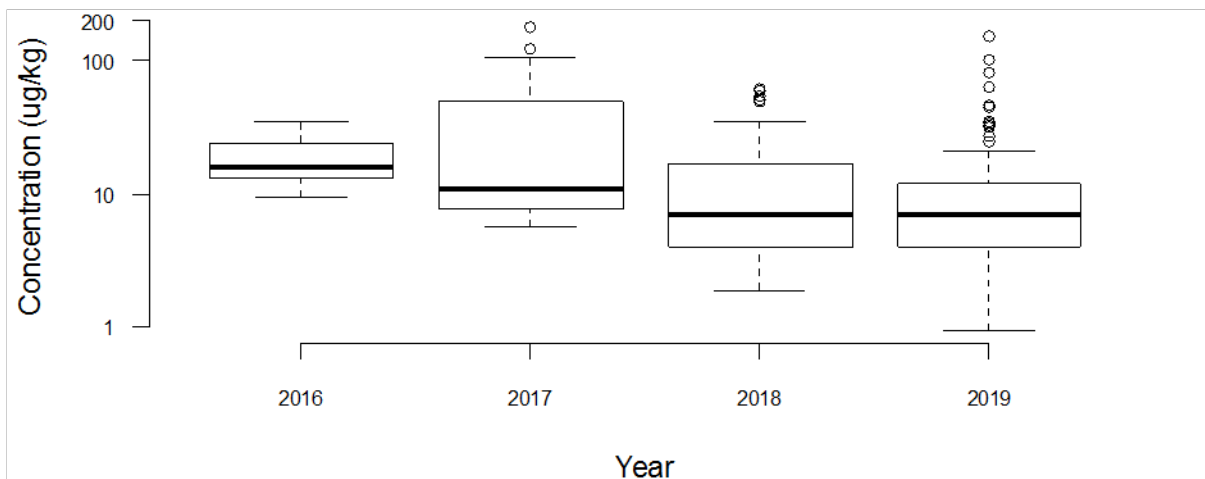
Because the data do not meet the minimum requirements for trend reporting, a formal trends assessment is not reported. The blank entry on the dashboard indicates, in part, that there are insufficient data.

4.15.4 Thresholds

There appear to be no suitable threshold measures available for lead and cadmium concentrations in whole fish. The blank entries in the dashboard, in part, reflect that are no values available for comparison.

4.16 Heavy metals in Eurasian otter (*Lutra lutra*): lead and cadmium

Lead   No dashboard entry – insufficient or no comparable data

Cadmium

4.16.1 Data source

Data on lead and cadmium in otter livers have been provided by the CUOP (Cardiff University, 2020). Livers have been collected from individuals found dead each year. Most animals died as a result of traffic collisions but some individuals died from other causes.

4.16.2 Data structure

The data consist of a variable number of measurements of lead and cadmium concentrations in liver made each year between 2007 and 2016, excluding years 2009 to 2013, from a stratified random sub-sample[8] of all animals collected from England and Wales. Data for samples only from England are considered in this report.

Samples were analysed in three main tranches that included the years 2006–2008, 2009 (cadmium only) and 2014–2016. However, the LoD for lead for the first two tranches of data (0.12µg/g dry weight) was an order of magnitude higher compared with that for the last tranche (0.016µg/g dry weight). The data are presented as generated, with samples with concentrations below the LoD assigned half their respective LoD values (0.06µg/g dry weight and 0.008µg/g dry weight), but analysis of change over time was restricted to data from years 2014–2016.

The LoD for cadmium also varied between analytical batches, but to a lesser extent, and few individuals (4 out of 154 otters across all years) had cadmium concentrations in liver below the LoD. Therefore, the highest LoD (0.012µg/g dry weight) was simply applied to the whole cadmium dataset, and a value of half the LoD (0.006µg/g dry weight) was assigned to non-detected concentrations.

The data for lead and cadmium concentrations in liver were not normally distributed for most years, either as measured concentrations or log10-transformed data; therefore, non-parametric descriptive statistics are reported here.

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[8] Overall sample set stratified so it was representative of age class, sex and year. Livers selected from animals at random from within those categories

4.16.3 Exploration of change in chemical concentrations over time

The distribution of data over time is shown in Figure 4.16.1 and is summarised in Table 4.16.1.

Figure 4.16.1 Scatterplots of lead and cadmium residues in the liver of Eurasian otters from England. Data shown are for individuals. Horizontal lines within plots indicate annual median values (diagram courtesy of UKCEH)
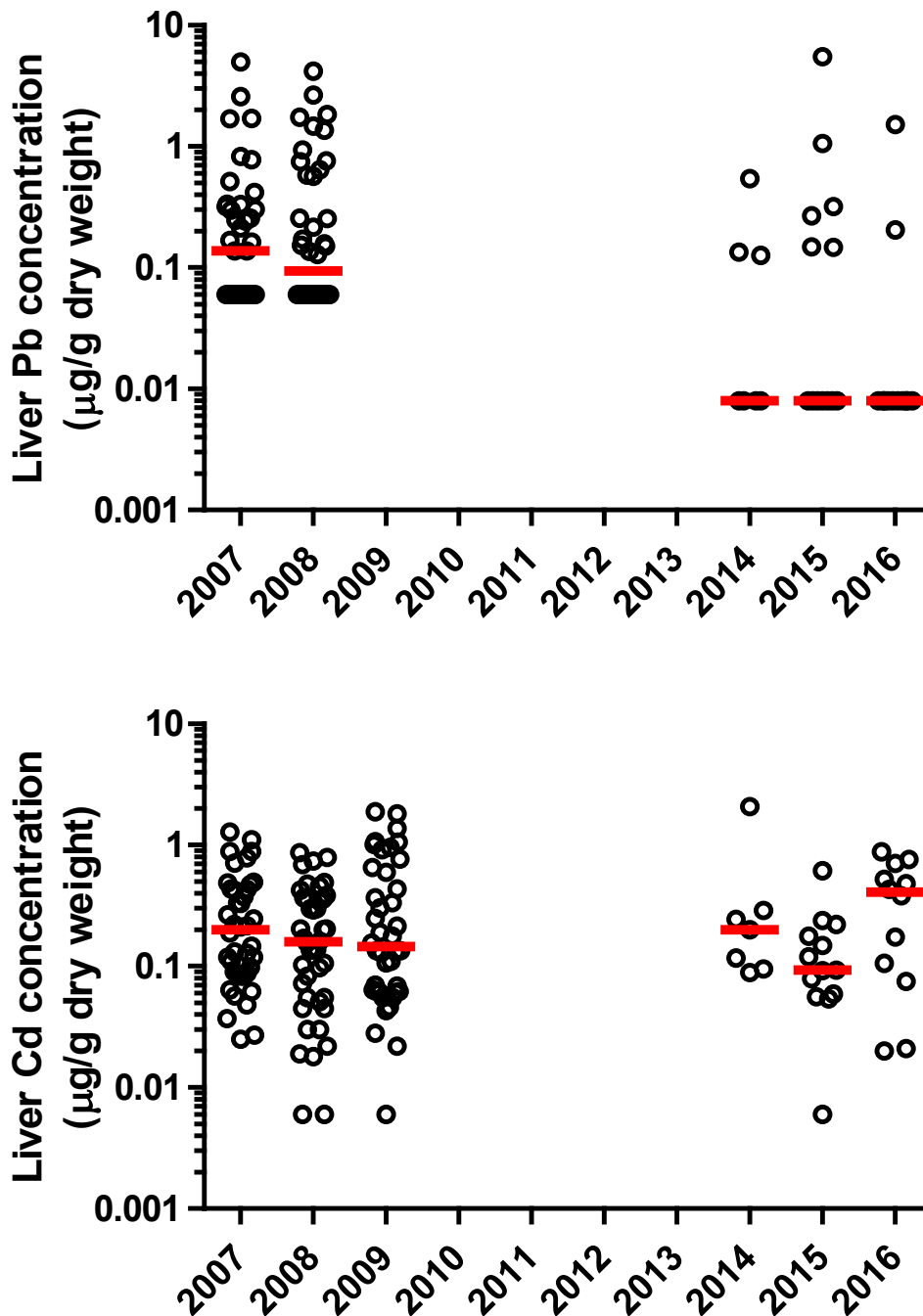
Table 4.16.1 Summary statistics for lead and cadmium concentrations in otter liver (µg/g dry weight)[1]

Substance	Year	n	Mean	SD	Median	Min	Max	Q1	Q3
Lead	2007	40	0.445	0.905	0.138	0.060	4.98	0.060	0.330
Lead	2008	42	0.486	0.837	0.094	0.060	4.20	0.060	0.598
Lead	2014	7	0.119	0.196	0.008	0.008	0.544	0.008	0.134
Lead	2015	13	0.578	1.51	0.008	0.008	5.51	0.008	0.295
Lead	2016	12	0.150	0.435	0.008	0.008	1.52	0.008	0.008
Cadmium	2007	40	0.310	0.313	0.201	0.025	1.28	0.087	0.435
Cadmium	2008	42	0.238	0.227	0.159	0.006	0.869	0.054	0.361
Cadmium	2009	40	0.400	0.497	0.145	0.006	1.89	0.062	0.639
Cadmium	2014	7	0.445	0.725	0.200	0.089	2.08	0.095	0.290
Cadmium	2015	13	0.151	0.155	0.093	0.006	0.615	0.058	0.200
Cadmium	2016	12	0.380	0.303	0.408	0.020	0.88	0.083	0.658

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value; Pb: lead; Cd: cadmium.

Visual inspection of the data (Figure 4.16.1) suggests that lead concentrations in liver may have declined over the whole time period. However, there was no statistically significant difference, based on a Kruskal–Wallis test (KW = 2.27, p = 0.32), in concentrations between 2014, 2015 and 2016, the most recent years for which there are data and for which the LoD is the same. However, given the paucity of data available for lead, the entry in the dashboard is blank.

There was no consistent trend in cadmium concentrations in otter livers and no statistically significant difference between any of the years (KW = 4.79, p = 0.44). For the dashboard, 'No observed change in concentrations' (↔) is applied for cadmium in otter liver.

4.16.4 Thresholds

There are no established threshold concentrations for lead or cadmium in otter livers and so no threshold value is proposed for this metric. The entries in the dashboard reflect that there are no values available for comparison.

4.17 Heavy metals in marine waters: lead and cadmium
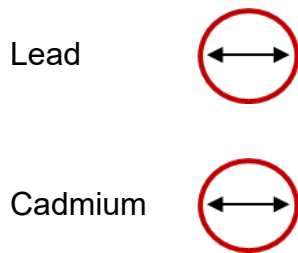
Lead

Cadmium

4.17.1 Data source

Data on dissolved lead and cadmium concentrations in estuarine and coastal waters have been provided by the Environment Agency from their statutory monitoring network.

4.17.2 Data structure

Relevant data are available for the period 2014–2019 for dissolved lead and cadmium from samples collected around the English coast. The data vary both in terms of the number of measurements taken within a year per site and the number of sites monitored per year. Some sites have been sampled in multiple years.

A data summary is available for each year based on the total number of measurements made in a year – that is all data pooled from all sites (see Tables 4.17.1 and 4.17.2). Summaries are also available for each site based on samples taken over the most recent 3 years and for which there were more than 3 samples per year.

For those samples reported as below the LoD, the less than values ranged from <0.04 to <0.2μg/L for dissolved lead and <0.03 and <0.3μg/L for dissolved cadmium. Half LoD values were assigned to non-detected samples in the treatment of the data. The large majority of these cases had LoDs at the lower end of the ranges given above.

Monitoring sites have been continuously reviewed, removing sites that do not show contamination issues and including new ones where a potential source of contamination has been newly identified. For this reason we have selected the period 2014–2019 for reporting here to maintain the integrity of the time series, although data are available prior to that.

4.17.3 Exploration of change in chemical concentrations over time

A summary of how dissolved lead and cadmium concentrations in saline waters around the English coast have varied over time is presented in Tables 4.17.1 and 4.17.2 and Figures 4.17.1 and 4.17.2, respectively.

The presentation of data in the figures differs to that for some of the other data sources in this report because the large number of samples or even sites for which data are available would otherwise result in a cluttered figure. For the purposes of clarity, the data in Figures

4.17.1 and 4.17.2 are presented as annual median, interquartile range and 10–90th percentiles of individual sample concentrations.

For dissolved lead, there are a high number of detections (61% of samples); simple visual inspection of the data indicates that the average values are fairly similar throughout the years examined (Table 4.17.1 and Figure 4.17.1). The majority of results for cadmium are below the LoD, but the situation varies year on year (Table 4.17.2 and Figure 4.17.2).

Table 4.17.1 Summary statistics for concentrations of dissolved lead in estuarine and coastal waters (µg/L)[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2014	154	1069	0.10	0.21	0.050	0.020	2.7	0.020	0.093
2015	152	1030	0.16	0.48	0.054	0.020	7.9	0.020	0.12
2016	175	1216	0.13	0.30	0.046	0.020	5.0	0.020	0.098
2017	160	1200	0.091	0.18	0.042	0.020	2.5	0.020	0.088
2018	155	1031	0.13	0.34	0.057	0.020	7.3	0.020	0.12
2019	156	949	0.18	0.71	0.049	0.020	16	0.020	0.11

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Table 4.17.2 Summary statistics for concentrations of dissolved cadmium in estuarine and coastal waters (µg/L)[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2014	162	1151	0.023	0.027	0.015	0.015	0.48	0.015	0.015
2015	159	1098	0.031	0.028	0.015	0.015	0.15	0.015	0.038
2016	181	1289	0.023	0.019	0.015	0.015	0.16	0.015	0.015
2017	170	1273	0.026	0.022	0.015	0.015	0.19	0.015	0.032

| Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2018	159	1097	0.032	0.030	0.015	0.015	0.38	0.015	0.041
2019	158	1008	0.023	0.026	0.015	0.015	0.46	0.015	0.015

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Figure 4.17.1 Median, interquartile range and 10–90th percentiles of dissolved lead concentrations in estuarine and coastal waters (µg/L) for samples taken from all sites
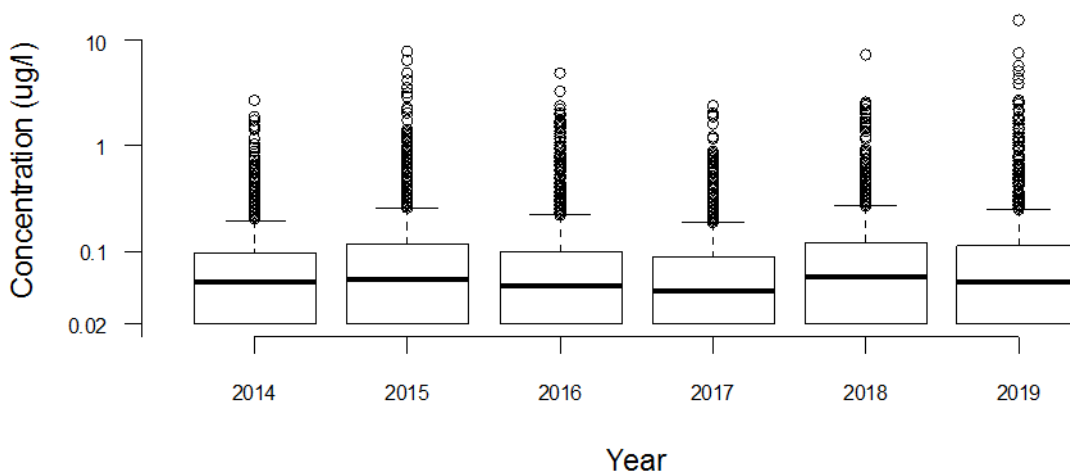
Figure 4.17.2 Median, interquartile range and 10–90th percentiles of dissolved cadmium concentrations in estuarine and coastal waters (µg/L) for samples taken from all sites
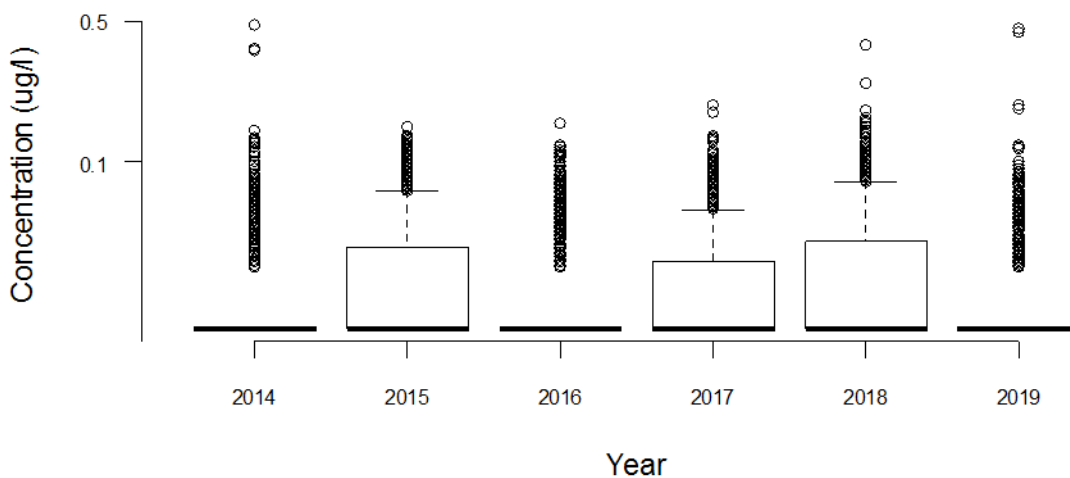
For the purposes of providing national trend assessment data for the dashboard, all sample data were considered in the temporal trend analysis. The geometric mean was taken of all samples at all sites per year for each metal to minimise any skews in the data

and the undue influence of outliers. Seasonal patterns were considered to minimise the impact of quality varying throughout the year owing to natural cycles; measurements were grouped into four seasons: winter (January to March), spring (April to June), summer (July to September), and autumn (October to December). The tseries package in R was used to analyse the time series and identify any potential trends, the significance of which was assessed using the Cox Stuart trend test. Statistically significant trends were those for which the p-value was <0.05.

Assessments based on medians and geomeans of sites and were also performed but there was no difference in the overall conclusions.

Figures 4.17.3 and 4.17.4 show the breakdown of the time series data for lead and cadmium, respectively, based on seasonal geometric means of all samples from all sites. In each figure, there are four graphs illustrating the raw data as a seasonal value ('data'), the effects of the seasonal pattern (repeating the four seasons for each year; 'seasonal'), the trend ('trend'), and the residual concentrations once the seasonal and trend series data are removed ('remainder').

Assessment of the data shows that there are no statistically significant trends observed based on the geometric means for dissolved lead and cadmium (Table 4.17.3). The dashboard trend information is based on the overall national assessment; therefore, the corresponding entry is 'No observed change in concentrations'.

Figure 4.17.3 Breakdown of trend analysis of seasonal geometric means of dissolved lead (µg/L) in estuarine and coastal waters
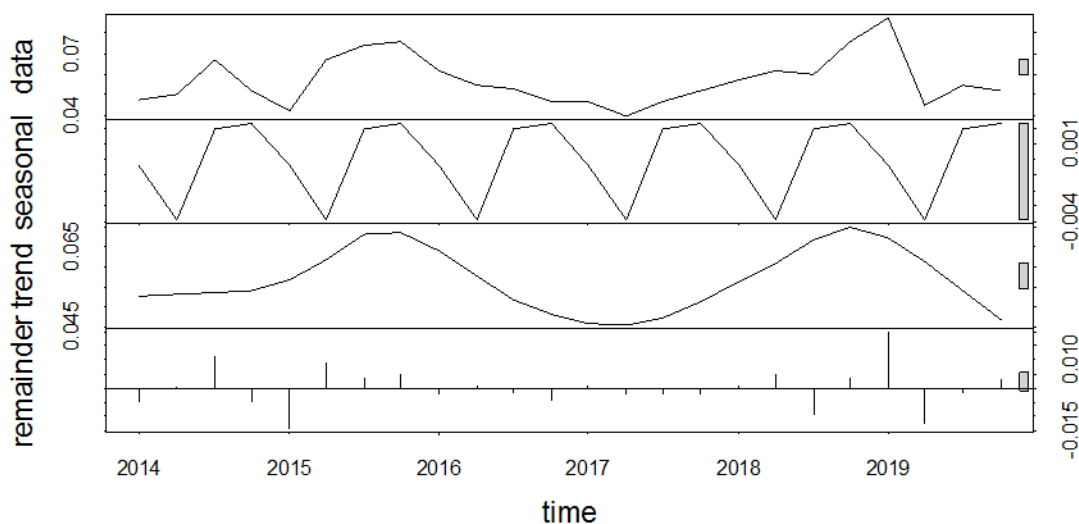
Figure 4.17.4 Breakdown of trend analysis of seasonal geometric means of dissolved cadmium (μg/L) in estuarine and coastal waters
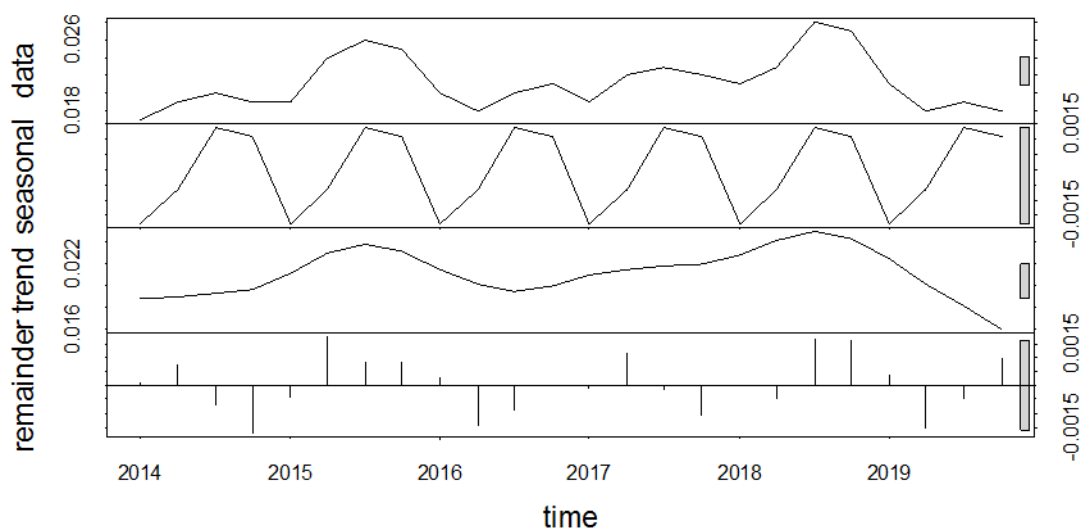
Table 4.17.3 Summary of p-values from the temporal trend assessment of the geometric means of bioavailable lead and dissolved cadmium concentrations in estuarine and coastal waters

Substance	p-value any trend	p-value downward trend	p-value upward trend	Decision
Lead	0.75	0.38	0.83	no change
Cadmium	0.23	0.97	0.11	no change

4.17.4 Thresholds

Threshold values applied to the data were annual average EQS values for dissolved lead (1.3μg/L) and dissolved cadmium (0.2μg/L) for such waters (UK Government, 2015).

Typically, average site concentrations are used for comparison with the EQSs for lead and cadmium. These are based on available data for a 3-year period. The assessment here is based on site averages for the period 2017–2019. Not every site has the maximum number of years' data available; over three-quarters of sites had data for one year. Each site requires >3 samples taken over that period to be included in the assessment; the number of samples per site varied between 5 and 36 for lead and 5 and 46 for cadmium.

The number and proportion of sites with mean concentrations for 2017–2019 that exceed the above thresholds have been calculated. No sites had average values above the EQSs for dissolved lead or cadmium and these results are used for the corresponding entries in the dashboard.

4.18 Heavy metals in marine waters: nickel, copper and zinc
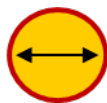
Nickel

Copper

Zinc

4.18.1 Data source

Data on dissolved nickel, copper and zinc concentrations in estuarine and coastal waters have been provided by the Environment Agency from their statutory monitoring network.

4.18.2 Data structure

Relevant data are available for the period 2014–2019 for dissolved nickel, copper and zinc from samples collected around the English coast. The data vary both in terms of the number of measurements taken within a year per site and the number of sites monitored per year. Some sites have been sampled in multiple years.

A data summary is available for each year based on the total number of measurements made in a year – that is all data pooled from all sites (see Tables 4.18.1, 4.18.2 and 4.18.3). Summaries are also available for each site based on samples taken over the most recent 3 years and for which there were more than 3 samples per year.

For those samples reported as below the LoD, the less than values were predominantly <0.3µg/L for dissolved nickel, <0.2µg/L for dissolved copper and <0.4µg/L for dissolved zinc. Half LoD values were assigned to non-detected samples in the treatment of the data.

Monitoring sites have been continuously reviewed, removing sites that do not show contamination issues and including new ones where a potential source of contamination has been newly identified. For these reasons we have selected the period 2014–2019 for reporting here to maintain the integrity of the time series, although data are available prior to that.

4.18.3 Exploration of change in chemical concentrations over time

A summary of how dissolved nickel, copper and zinc concentrations in saline waters around the English coast have varied over time is presented in Tables 4.18.1, 4.18.2 and 4.18.3 and Figures 4.18.1, 4.18.2 and 4.18.3, respectively.

The presentation of data in the figures differs to that for some of the other data sources in this report because the large number of samples or even sites for which data are available would otherwise result in a cluttered figure. For the purposes of clarity, the data in Figures 4.18.1, 4.18.2 and 4.18.3 are presented as annual median, interquartile range and 10–90th percentiles of individual sample concentrations.

Table 4.18.1 Summary statistics for concentrations of dissolved nickel in estuarine and coastal waters (μg/L)[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2014	156	1102	1.2	1.2	0.80	0.15	14	0.48	1.5
2015	154	1062	1.2	1.1	0.80	0.15	11	0.45	1.6
2016	177	1262	1.1	1.1	0.85	0.15	15	0.45	1.6
2017	165	1250	1.1	1.1	0.78	0.15	20	0.48	1.4
2018	157	1080	1.6	8.0	1.1	0.15	260	0.52	2.0
2019	159	997	1.2	1.2	0.79	0.15	14	0.47	1.6

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Table 4.18.2 Summary statistics for concentrations of dissolved copper in estuarine and coastal waters (μg/L)[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2014	178	614	1.4	1.2	1.1	0.10	12	0.67	1.7
2015	171	488	1.4	1.0	1.2	0.10	5.1	0.68	2.0
2016	190	677	1.3	0.95	1.1	0.10	5.2	0.59	1.9
2017	181	698	1.5	1.1	1.2	0.10	7.4	0.68	2.1
2018	169	490	1.6	1.3	1.3	0.10	10	0.73	2.3

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2019	168	438	1.5	1.2	1.1	0.10	5.9	0.63	1.9

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Table 4.18.3 Summary statistics for concentrations of dissolved zinc in estuarine and coastal waters (µg/L)[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2014	169	568	3.5	2.9	2.7	0.20	20	1.5	4.3
2015	168	467	4.3	6.2	3.2	0.20	116	1.7	5.0
2016	188	660	4.0	4.0	2.7	0.20	20	1.2	4.7
2017	178	682	4.5	4.6	2.7	0.20	22	1.3	5.7
2018	172	508	4.7	4.4	3.3	0.20	26	1.6	6.2
2019	171	460	3.8	4.3	2.6	0.20	57	1.2	4.8

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Figure 4.18.1 Median, interquartile range and 10–90th percentiles of dissolved nickel concentrations in estuarine and coastal waters (µg/L) for samples taken from all sites
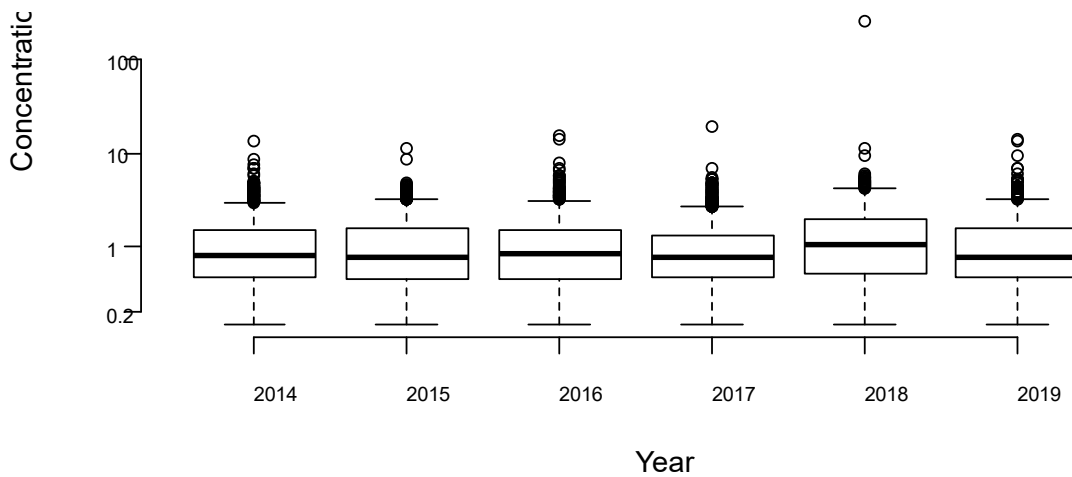
Figure 4.18.2 Median, interquartile range and 10–90th percentiles of dissolved copper concentrations in estuarine and coastal waters (µg/L) for samples taken from all sites
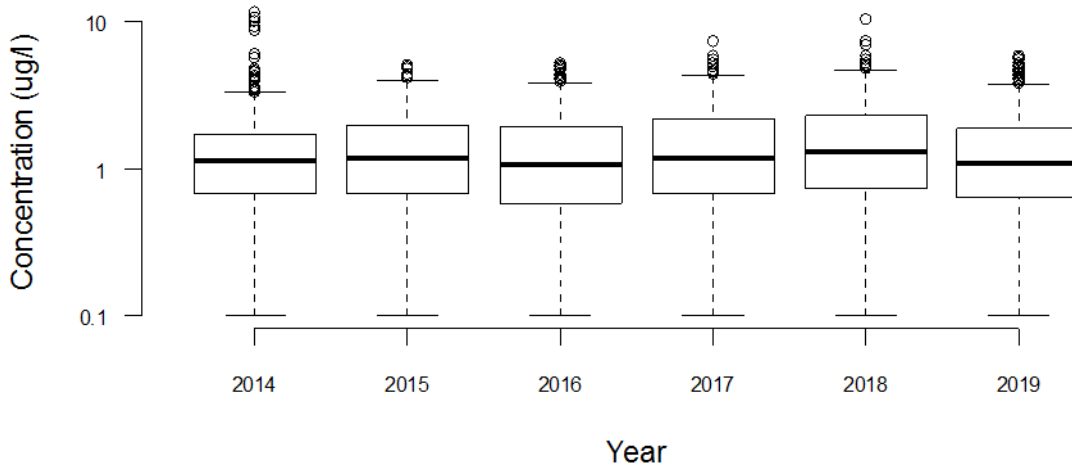
Figure 4.18.3 Median, interquartile range and 10–90th percentiles of dissolved zinc concentrations in estuarine and coastal waters (µg/L) for samples taken from all sites
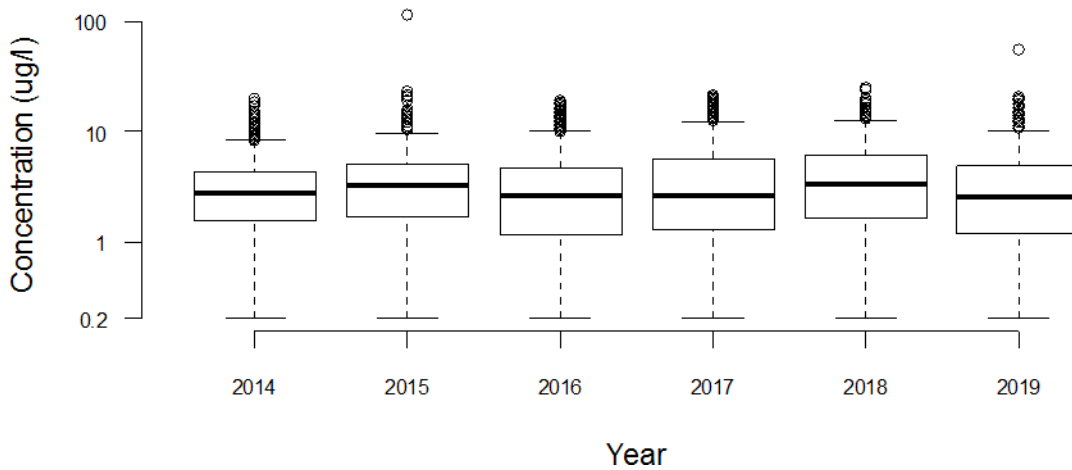
For the purposes of providing national trend assessment data for the dashboard, all sample data were considered in the temporal trend analysis. The same method was used as described in Section 4.17.3.

Assessments based on medians and geomeans of sites and were also performed but there was no difference in the overall conclusions.

Figures 4.18.4, 4.18.5 and 4.18.6 show the breakdown of the time series data for lead and cadmium, respectively, based on seasonal geometric means of all samples from all sites. In each figure, there are four graphs illustrating the raw data as a seasonal value ('data'), the effects of the seasonal pattern (repeating the four seasons for each year; 'seasonal'), the trend ('trend'), and the residual concentrations once the seasonal and trend series data are removed ('remainder').

Figure 4.18.4 Breakdown of trend analysis of seasonal geometric means of dissolved nickel (µg/L) in estuarine and coastal waters
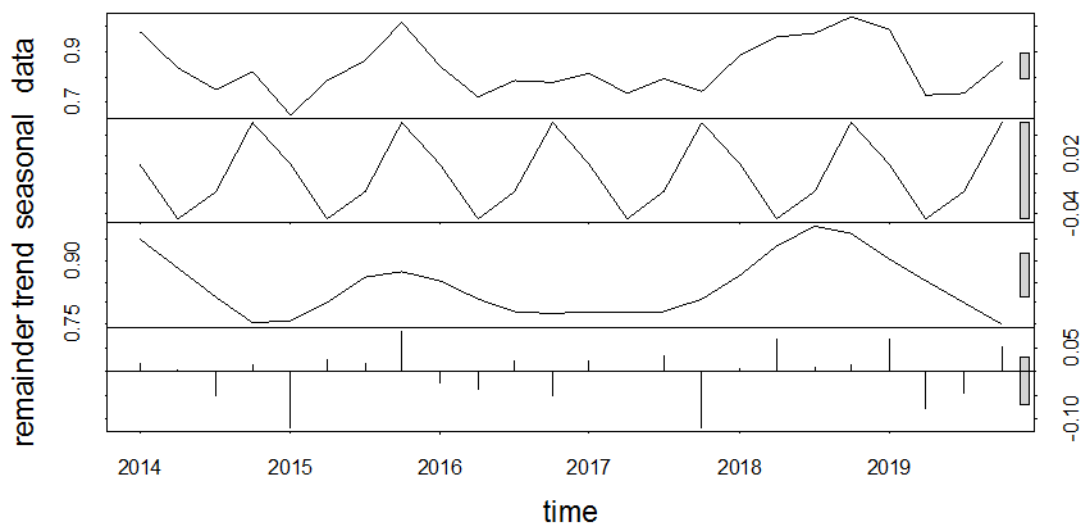
Figure 4.18.5 Breakdown of trend analysis of seasonal geometric means of dissolved copper (µg/L) in estuarine and coastal waters
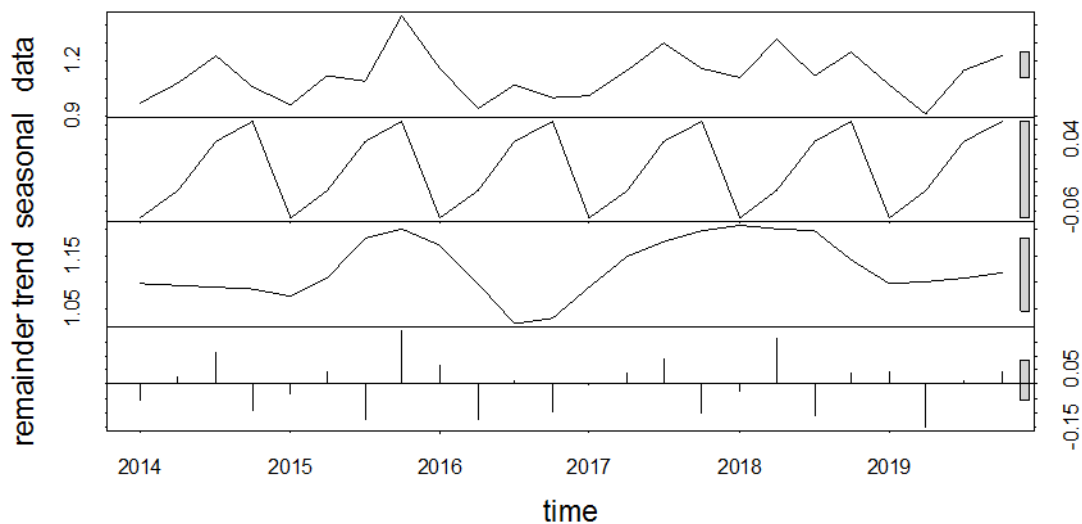
Figure 4.18.6 Breakdown of trend analysis of seasonal geometric means of dissolved zinc (μg/L) in estuarine and coastal waters
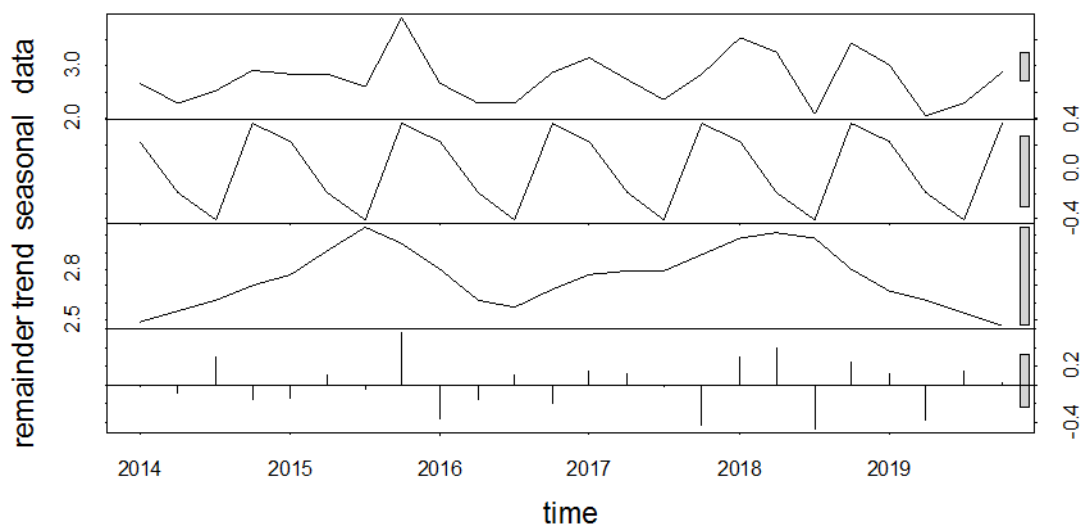
Assessment of the data shows that there are no statistically significant trends observed based on the geometric means for dissolved nickel, copper and zinc (Table 4.18.4). The dashboard trend information is based on the overall national assessment; therefore, the corresponding entry is 'No observed change in concentrations'.

Table 4.18.4 Summary of p-values from the temporal trend assessment of the geometric means of dissolved nickel, copper and zinc concentrations in estuarine and coastal waters

Substance	p-value any trend	p-value downward trend	p-value upward trend	Decision
Nickel	0.39	0.93	0.19	no change
Copper	0.15	0.98	0.073	no change
Zinc	1.0	0.61	0.61	no change

4.18.4 Thresholds

An annual average EQS value of 8.6μg/L for dissolved nickel in surface waters other than inland ones is given in the Water Framework Directive (Standards and Classification) Directions 2015 (UK Government, 2015). For copper and zinc, EQSs are also specified in the 2015 Directions for England and Wales (UK Government, 2015):

For dissolved copper, the EQS varies depending on DOC content at the sampling site: where DOC ≤1mg/L, the EQS is 3.76μg/L; where DOC >1mg/L, the EQS is 3.76 + (2.677 x ((DOC/2) – 0.5)) μg/L. This accounts for the fact that the ecotoxicity of copper has been shown to significantly reduce with increasing DOC (Maycock et al., 2012).

For zinc, the EQS is 6.8µg/L over and above any ABC present ([UKTAG, 2013](#)). A saline ABC of 1.1µg/L was added to the EQS to give a threshold of 7.9µg/L for use as a comparison here.

Typically, average site concentrations are used for comparison with the EQSs for these metals. These are based on available data for a 3-year period. The assessment here is based on site averages for the period 2017–2019. Not every site has the maximum number of years' data available, although typically just under two-thirds of the sites had data for the 3 years. Each site requires >3 samples taken over that period to be included in the assessment; the number of samples per site varied between 5 and 46 for nickel and 4 and 34 for copper and zinc.

The number and proportion of sites with mean concentrations for 2017–2019 that exceed the above thresholds have been calculated. For nickel, only 1 out of 209 sites (0.5%) had a mean concentration above the threshold of 8.6µg/L. For copper, none of the 74 sites sampled (0%) was above the corresponding EQSs. Zinc had the highest rate of exceedance with 17 out of 72 sites (24%) exceeding the threshold. The percentage results are used for the corresponding entries in the dashboard.

4.19 Heavy metals in blue mussels (*Mytilus edulis*): lead and cadmium
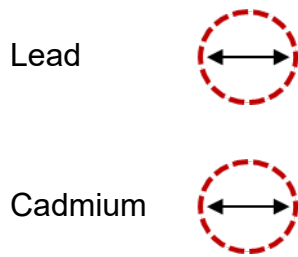
Lead

Cadmium

4.19.1 Data source

Data on lead and cadmium in blue mussels (*Mytilus edulis*) in England have been provided by the Environment Agency. Concentration data in *Mytilus* flesh have been collected, since around 2000, as part of the UK-wide OSPAR CEMP, with analysis later expanded in anticipation of monitoring requirements under the Water Environment Regulations 2017 (UK Government, 2017).

Data used in this assessment are also submitted, as part of the wider UK dataset, to the DOME (marine environment) data portal for the ICES (ICES, 2021a).

4.19.2 Data structure

Data on lead and cadmium concentrations in *Mytilus* flesh are available for the period 2000–2019, but only data from 2011 are included in this assessment to eliminate the impacts of historical changes on the monitoring programme.

The monitoring methodology is described in the CEMP programme manual, the Green Book (BODC, 2020). Where feasible sites are monitored annually, with a target of three samples – consisting of pooled individuals – collected at each site on each sampling occasion. Samples are collected in the winter/early spring, to avoid any seasonal influence from spawning.

Data are reported as µg/kg wet weight. All reported lead and cadmium concentrations were above the LoD.

4.19.3 Exploration of change in chemical concentrations over time

Data have been summarised for each year based on the total number of measurements made in a year – that is all data pooled from all sites (Tables 4.19.1 and 4.19.2, respectively) – and are also shown in Figures 4.19.1 and 4.19.2, respectively.

Temporal trend was assessed using the data for all individual samples analysed for lead and cadmium. The geometric mean was taken of all samples at all sites per year for each metal to minimise any skews in the data and the undue influence of outliers. The tseries package in R was used to analyse the time series and identify any potential trends, the

significance of which was assessed using the Cox Stuart trend test. Statistically significant trends were those for which the p-value was <0.05.

Assessments based on medians and geomeans of sites and were also performed but there was very little difference in the results.

Table 4.19.1 Summary data for lead in *Mytilus edulis* (μg/kg wet weight) from samples taken at monitored sites in England since 2011[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2011	17	47	691	507	516	176	2010	339	853
2012	17	33	686	378	593	249	1750	383	930
2013	17	51	593	409	497	247	2750	337	745
2014	20	61	772	734	517	258	3990	363	834
2015	19	57	549	403	393	180	1940	318	620
2016	16	48	738	623	472	250	2790	378	707
2017	15	45	687	502	473	261	1890	346	797
2018	13	37	499	310	396	216	1290	300	547
2019	16	46	552	369	431	171	2030	310	710

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; median; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Table 4.19.2 Summary data for cadmium in *Mytilus edulis* (µg/kg wet weight) from samples taken at monitored sites in England since 2011[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2011	17	47	198	115	144	84	447	107	278
2012	17	33	218	121	181	78	531	138	292
2013	17	51	153	100	120	73	498	94	158
2014	20	61	163	83	142	78	434	116	167
2015	19	57	209	138	148	76	633	122	242
2016	16	48	243	200	147	81	969	114	278
2017	15	45	191	112	130	70	473	103	244
2018	13	37	186	156	104	66	618	86	243
2019	16	46	199	151	158	69	665	106	212

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; median; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Figure 4.19.1 Median, interquartile range and 10–90th percentiles of concentrations of lead in *Mytilus edulis* (µg/kg wet weight) since 2011
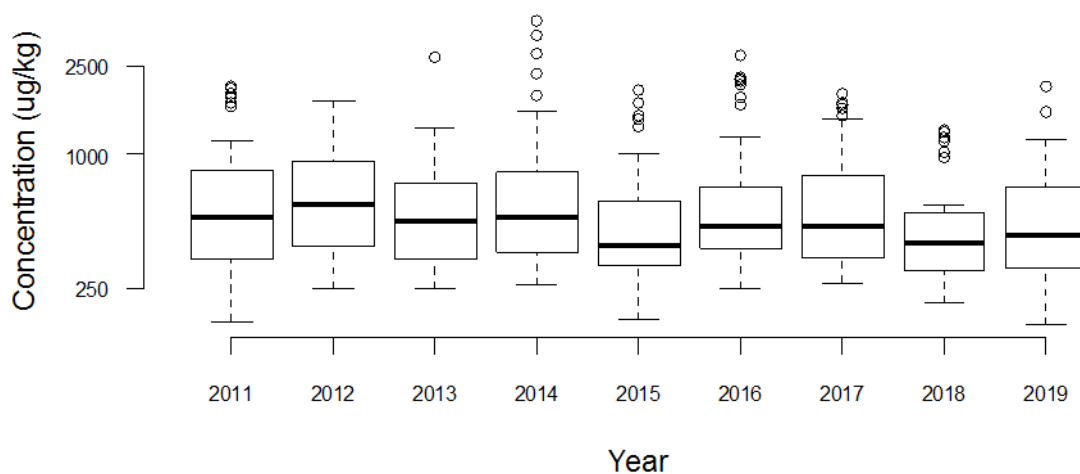
Figure 4.19.2 Median, interquartile range and 10–90th percentiles of concentrations of cadmium in *Mytilus edulis* (μg/kg wet weight) since 2011
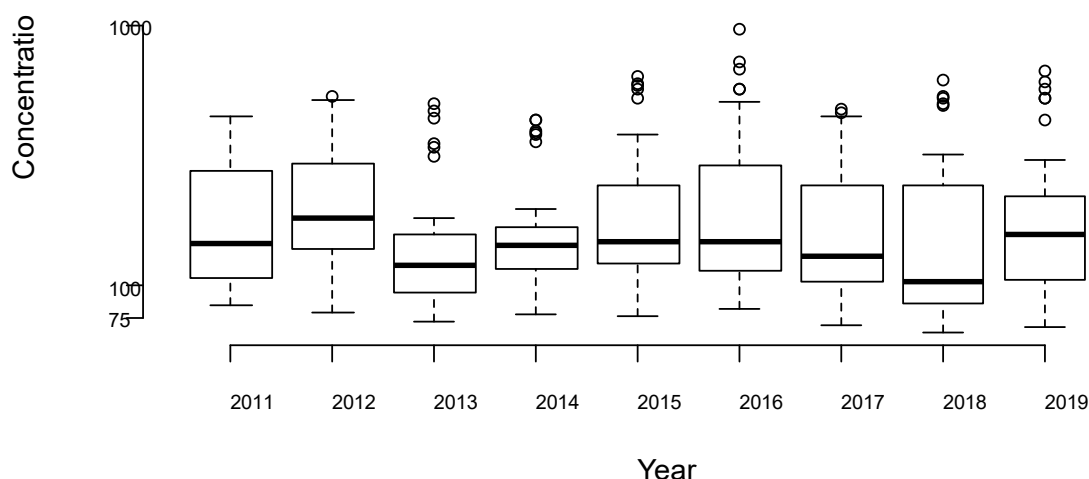
There was no overall temporal trend found for lead or cadmium in *Mytilus* flesh (Table 4.19.3). Therefore, the corresponding entries in the dashboard are for 'No observed change in concentrations'.

Table 4.19.3 Summary of p-values from the temporal trend assessment of the annual geometric means of lead and cadmium concentrations in *Mytilus edulis*

Substance	p-value any trend	p-value downward trend	p-value upward trend	Decision
Lead	0.13	0.063	1.0	no change
Cadmium	1.0	0.69	0.69	no change

4.19.4 Thresholds

There are currently no established thresholds for lead or cadmium in *Mytilus* under OSPAR or derived EQSs The entries in the dashboard reflect that there are no values available for comparison.

4.20 Heavy metals in blue mussels (*Mytilus edulis*): nickel, copper and zinc
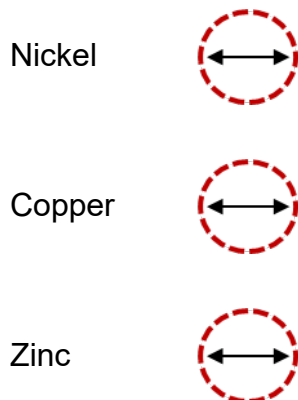
Nickel

Copper

Zinc

4.20.1 Data source

Data on nickel, copper and zinc in blue mussels (*Mytilus edulis*) in England have been provided by the Environment Agency. Concentration data in *Mytilus* flesh have been collected, since around 2000, as part of the UK-wide OSPAR CEMP.

Data used in this assessment are also submitted, as part of the wider UK dataset, to the DOME (marine environment) data portal for the ICES (ICES, 2021a).

4.20.2 Data structure

Data on nickel, copper and zinc concentrations from *Mytilus* flesh are available for the period 2000–2019, but only data from 2011 are included in this assessment to eliminate the impacts of historical changes on the monitoring programme.

The monitoring methodology is described in the CEMP programme manual, the Green Book (BODC, 2020). Where feasible sites are monitored annually, with a target of three samples – consisting of pooled individuals – collected at each site on each sampling occasion. Samples are collected in the winter/early spring, to avoid any seasonal influence from spawning.

Data are reported as µg/kg wet weight. The analytical LoD available for nickel is approximately 40µg/kg wet weight; however, 16% of samples were recorded at an elevated minimum reporting value of <300µg/kg wet weight and their results were set to half the face value. For copper, the analytical LoD is approximately 100µg/kg wet weight. However, a few samples (0.5%) had an elevated minimum reporting value of <800µg/kg wet weight and these results were also set to half the face value. All zinc data were above the LoD and there were no results with a less than (<) qualifier.

4.20.3 Exploration of change in chemical concentrations over time

The nickel, copper and zinc data have been summarised for each year based on the total number of measurements made in a year – that is all data pooled from all sites (Tables 4.20.1, 4.20.2 and 4.20.3, respectively). The data are also shown in Figures 4.20.1, 4.20.2 and 4.20.3, respectively.

Table 4.20.1 Summary data for nickel in *Mytilus edulis* (µg/kg wet weight) from samples taken at monitored sites in England since 2011[1]

| Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2011	17	47	245	132	150	150	606	150	323
2012	17	33	439	262	370	150	1340	261	529
2013	17	51	328	145	304	153	690	205	435
2014	20	61	674	444	624	150	2570	405	844
2015	19	57	264	137	150	150	589	150	345
2016	16	48	342	195	327	150	767	150	505
2017	15	45	353	113	350	151	584	261	454
2018	13	37	410	122	398	230	659	299	499
2019	16	46	294	114	261	141	640	202	378

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; median; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value

Table 4.20.2 Summary data for copper in *Mytilus edulis* (μg/kg wet weight) from samples taken at monitored sites in England since 2011[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2011	17	47	1611	1608	1120	583	8890	938	1395
2012	17	33	1842	761	1570	967	3870	1350	2150
2013	17	51	1900	1872	1130	866	7450	991	1580
2014	20	61	2709	2244	2030	400	11400	1630	2550
2015	19	57	1185	312	1150	400	2250	990	1350
2016	16	48	1472	761	1255	836	4830	1008	1620
2017	15	45	1761	1544	1190	812	6490	952	1590
2018	13	37	1002	317	931	522	1670	773	1220
2019	16	46	1057	301	1004	540	1760	845	1135

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; median; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value

Table 4.20.3 Summary data for zinc in *Mytilus edulis* (μg/kg wet weight) from samples taken at monitored sites in England since 2011[1]

Year	Number of sites	n	Mean	SD	Median	Min	Max	Q1	Q3
2011	17	47	16492	4494	16400	8650	33800	13750	19300
2012	17	33	20203	6672	19400	10800	41700	15900	22100
2013	17	51	20020	10907	15600	10700	51300	13800	19750
2014	20	61	19761	6782	17400	12100	42500	15900	20900

2015	19	57	21368	9881	17000	8580	55300	15300	25600
2016	16	48	22713	9274	20700	9400	48000	16300	26475
2017	15	45	19453	7260	18500	8930	38500	14300	22200
2018	13	37	22157	13335	15300	9510	54000	13500	25700
2019	16	46	21296	10528	18050	11700	61500	16000	21800

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; median; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Figure 4.20.1 Median, interquartile range and 10–90th percentiles of concentrations of nickel in *Mytilus edulis* (µg/kg wet weight) since 2011
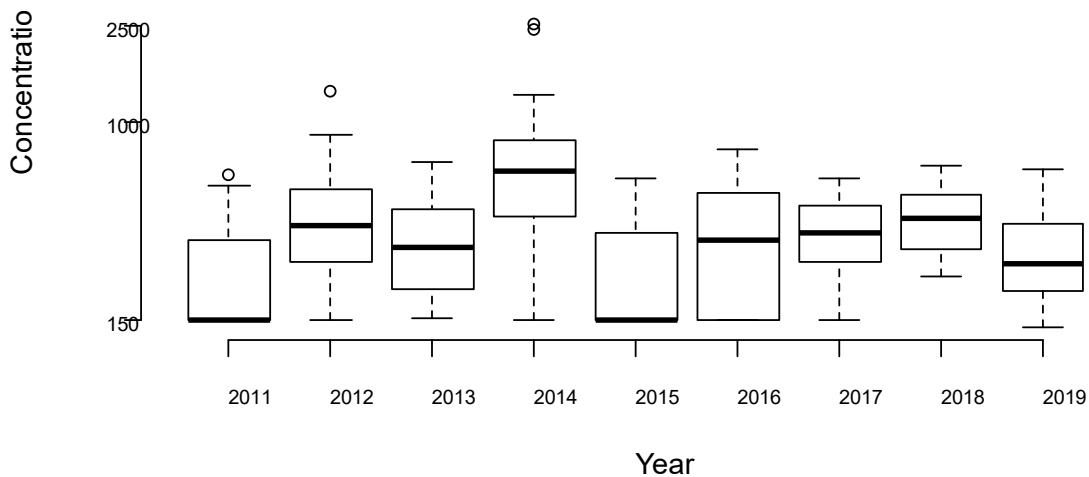
Figure 4.20.2 Median, interquartile range and 10–90th percentiles of concentrations of copper in *Mytilus edulis* (µg/kg wet weight) since 2011
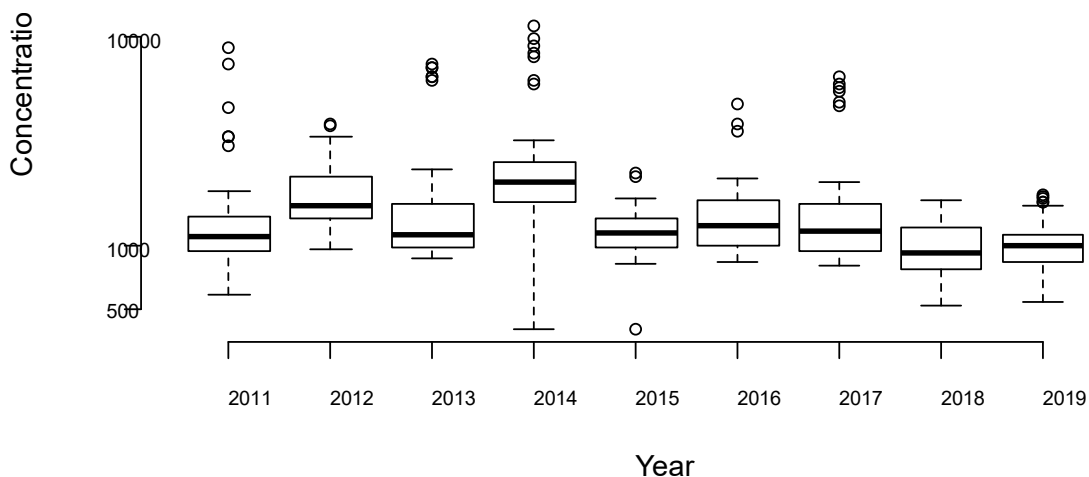
Figure 4.20.3 Median, interquartile range and 10–90 percentiles of concentrations of zinc in *Mytilus edulis* (µg/kg wet weight) since 2011
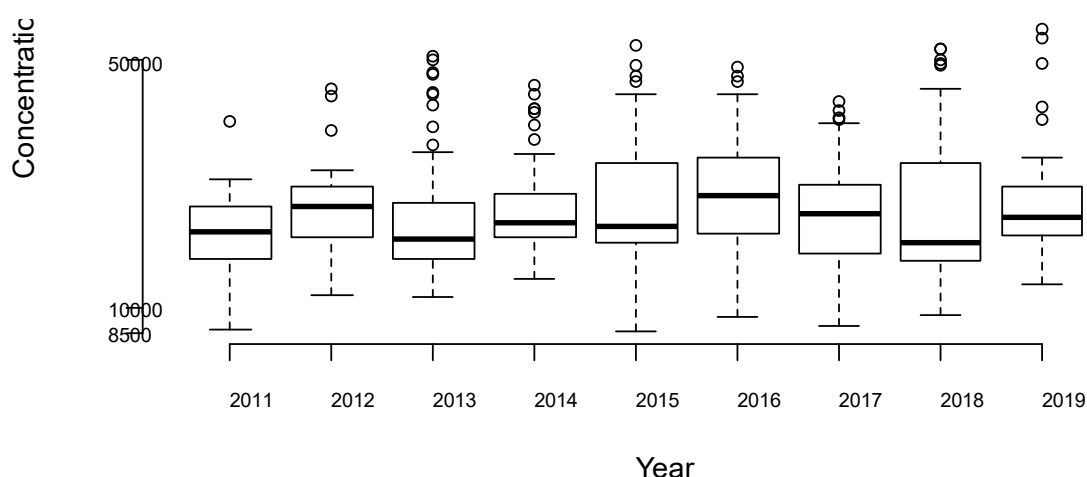
Temporal trend was assessed using the data for all individual samples analysed for nickel, copper and zinc. The data were converted into the geometric mean, to minimise any skews in the data and the undue influence of outliers. The tseries package in R was used to analyse the time series and identify any potential trends, the significance of which was assessed using the Cox Stuart trend test. Statistically significant trends were those for which the p-value was <0.05.

Assessments based on medians and geomeans of sites and were also performed but there was very little difference in the results.

There was no overall temporal trend found for nickel, copper or zinc in *Mytilus* flesh (Table 4.20.4). Therefore, the corresponding entries in the dashboard are for 'No observed change in concentrations'.

Table 4.20.4 Summary of p-values from the temporal trend assessment of the geometric means of nickel concentrations in *Mytilus edulis*

Substance	p-value any trend	p-value downward trend	p-value upward trend	Decision
Nickel	1.0	0.69	0.69	No change
Copper	0.63	0.31	0.94	No change
Zinc	1.0	0.69	0.69	No change

4.20.4 Thresholds

There are currently no established thresholds for nickel, copper and zinc in *Mytilus* under OSPAR or derived EQSs. The entries in the dashboard reflect that there are no values available for comparison.

4.21 Heavy metals in marine fish: lead and cadmium

Lead

Cadmium

4.21.1 Data source

Data on lead and cadmium concentrations in liver are available for dab (*Limanda limanda*). These data are collected as part of the Marine Strategy Framework Directive (UK Marine Strategy)–OSPAR (MSFD–OSPAR) monitoring for assessing good environmental status. The data are collected and held by Cefas.

Data used in this assessment are also submitted, as part of the wider UK dataset, to the DOME (marine environment) data portal for the ICES (ICES, 2021a).

Sites are selected on the basis that they reliably support dab populations that can be sampled for analysis, there are a minimum of three sites within each OSPAR hydro-geographical sub-region (OSPAR Commission, 2020) and there are no direct impacts from local sources.

Data have been provided for 2008–2019 covering 25 stations, although monitoring at one of these was only conducted in 2014. Between 2008 and 2010, sampling around the country was done annually and covered 16–23 stations each year. From 2011 onwards, fish were collected at east and west coast stations on alternate years. For these years, there are data for 14–15 east coast stations (odd years) and 8–9 west coast ones (even years).

All data relate to designated English waters, with the exception of those from a Welsh station in the Bristol Channel as this is a shared water body in which fish are likely to move freely across territorial water.

Typically, 5 or fewer pools of fish were sampled around each station. Each pool comprised 5 fish. Stations at which there was only one sample collected have been excluded from the analysis.

4.21.2 Data structure

Data are for lead and cadmium concentrations in liver. All concentration data are reported in mg/kg wet weight.

The data summaries that were provided consisted of results from individual pool samples taken around the stations. These individual samples were used in the trend and threshold assessments rather than station means for the purposes of the dashboard indicator. This

is because the pooled samples are already representative of a mean of 5 fish and this approach allows assessment of trends across the stations and nationally.

Within the datasets of 836 samples per substance, the LoDs ranged from 0.01–0.09 and 0.01–0.14mg/kg wet weight for lead and cadmium, respectively. Thirty-three samples for lead and 19 for cadmium were reported below those LoDs; these were assigned values equal to half the LoD.

One sample taken from a station in 2019 was removed as an extreme positive outlier for both metals.

4.21.3 Exploration of change in chemical concentrations over time

Summary data for lead and cadmium concentrations in dab liver from samples analysed across the period are given in Tables 4.21.1 and 4.21.2, respectively.

To bring the data as close to a normal distribution as possible, the measured concentrations were converted into Ln values for the purpose of assessing trends. Plots of the overall change in lead and cadmium Ln concentrations in dab liver from 2008 to 2019 are shown in Figure 4.21.1. For cadmium the plot seems to be cyclic and reflects the higher values seen on the east coast in 2013, 2015, 2017 and 2019 compared with those taken on alternate years on the west (Figure 4.21.1 and Table 4.21.2).

Table 4.21.1 Summary statistics for samples of lead in dab liver (mg/kg wet weight)[1]

Year	Number of stations	n	Mean	SD	Median	Min	Max	Q1	Q3
2008	23	114	0.22	0.33	0.10	0.015	2.5	0.070	0.23
2009	23	115	0.17	0.22	0.080	0.010	1.2	0.040	0.21
2010	16	79	0.20	0.33	0.10	0.010	2.4	0.040	0.20
2011	15	74	0.14	0.27	0.070	0.0050	2.0	0.033	0.10
2012	8	40	0.42	0.48	0.26	0.050	2.4	0.090	0.60
2013	15	74	0.20	0.28	0.070	0.020	1.2	0.050	0.21
2014	9	45	0.29	0.22	0.24	0.060	1.1	0.12	0.40
2015	14	66	0.29	0.39	0.10	0.020	1.6	0.050	0.25

Year	Number of stations	n	Mean	SD	Median	Min	Max	Q1	Q3
2016	8	37	0.26	0.29	0.13	0.050	1.5	0.080	0.33
2017	15	75	0.30	0.56	0.080	0.020	3.0	0.040	0.18
2018	9	45	0.23	0.28	0.14	0.047	1.8	0.091	0.26
2019	15	70	0.25	0.47	0.064	0.020	2.7	0.044	0.18

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Table 4.21.2 Summary statistics for samples of cadmium in dab liver (mg/kg wet weight)[1]

Year	Number of stations	n	Mean	SD	Median	Min	Max	Q1	Q3
2008	23	114	0.23	0.22	0.16	0.020	1.8	0.10	0.31
2009	23	115	0.23	0.22	0.15	0.0050	1.2	0.095	0.29
2010	16	79	0.27	0.33	0.21	0.020	2.00	0.10	0.31
2011	15	74	0.27	0.30	0.20	0.030	2.2	0.11	0.32
2012	8	40	0.14	0.084	0.13	0.030	0.40	0.068	0.18
2013	15	74	0.49	0.56	0.34	0.050	3.7	0.23	0.53
2014	9	45	0.13	0.089	0.11	0.040	0.38	0.060	0.17
2015	14	66	0.49	0.46	0.36	0.050	2.0	0.17	0.57
2016	8	37	0.17	0.013	0.13	0.030	0.68	0.080	0.19
2017	15	75	0.48	0.45	0.38	0.070	3.1	0.24	0.55

Year	Number of stations	n	Mean	SD	Median	Min	Max	Q1	Q3
2018	9	45	0.086	0.032	0.078	0.043	0.19	0.063	0.11
2019	15	70	0.43	0.51	0.24	0.041	2.6	0.17	0.36

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Figure 4.21.1 Scatterplots of Ln lead and cadmium residues in the liver of dab from marine waters around England. Data shown are for individual samples. The solid black line shows the trend from a generalised additive model as a function of time (diagram courtesy of Cefas)
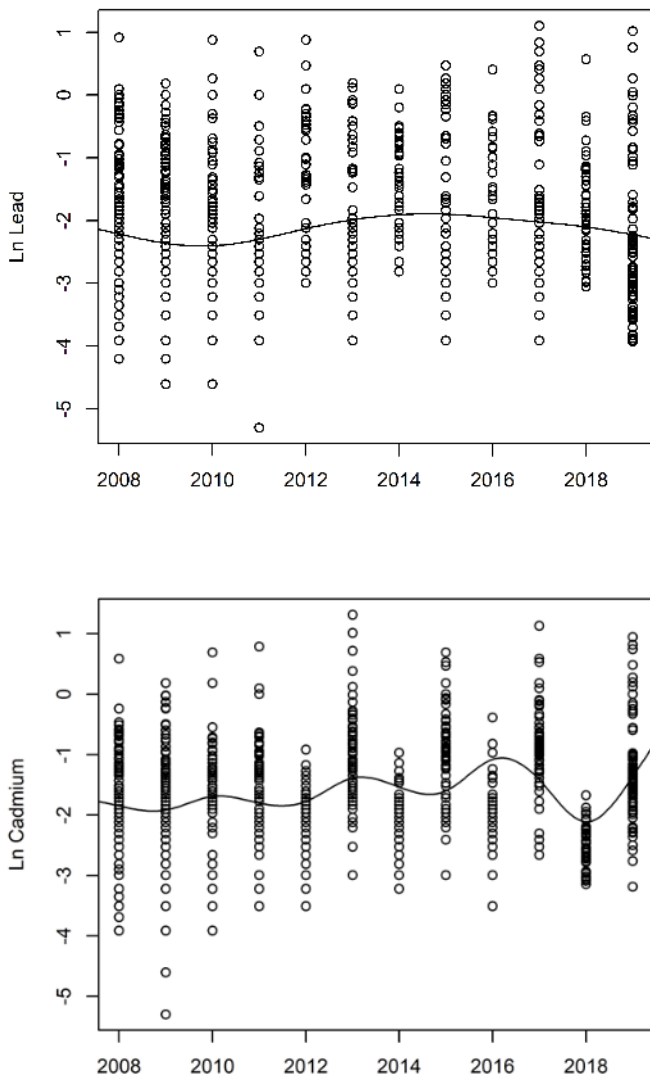
Minimum data requirements for trend assessment are met, though the monitoring regime has altered over time (see Section 4.21.1).

To determine the overall trend for the dashboard, temporal changes in concentrations at individual stations were assessed. This is because it is possible that analysing the results all together may give misleading conclusions for trends and mask intersite variations. Scatterplots of results for samples taken at these stations can be found in Appendix B, Figures B.4.21.1 and B.4.21.2 for lead and cadmium, respectively.

A GAM (Wood, 2017) was used for the assessment. The same criteria for interpreting the GAM plots for individual stations as described in Section 4.8.3 was used here. The statistic D for each metal was then calculated as also described in Section 4.8.3.

For lead, four stations, all on the west coast, show downward trends and 6 – 5 of these from the east coast – indicate upward trends (Figure B.4.21.1). The resulting D value (8%) strongly supports the assignment of no observed change (↔) and this is used within the dashboard.

Trends for cadmium residues in dab liver at individual stations do not show any major geographical split. Three stations are shown to have downward trends and six upward ones (Figure B.4.21.2). This results in a D value of 12% and an assignment of no observed change (↔); this is used within the dashboard.

4.21.4 Thresholds

There are no ecological thresholds either under the OSPAR framework or derived as statutory values. Therefore assessment of the most recent data against thresholds for lead and cadmium is not possible and the dashboard entries indicate that there are no corresponding thresholds defined.

It is worth noting the distinct difference between the east and west coast concentrations in dab liver for cadmium. Should a threshold become available in future, it may be worthwhile determining any geographical differences in threshold exceedance, alongside a national view, for this substance.

4.22 Heavy metals in marine fish: nickel, copper and zinc

Nickel

Copper

Zinc

4.22.1 Data source

Data on nickel, copper and zinc concentrations in liver are available for dab (*Limanda limanda*). These data are collected as part of the Marine Strategy Framework Directive (UK Marine Strategy)–OSPAR (MSFD–OSPAR) monitoring for assessing good environmental status. The data are collected and held by Cefas.

Data used in this assessment are also submitted, as part of the wider UK dataset, to the DOME (marine environment) data portal for the ICES (ICES, 2021a).

Sites are selected on the basis that they reliably support dab populations that can be sampled for analysis, there are a minimum of three sites within each OSPAR hydro-geographical sub-region (OSPAR Commission, 2020) and there are no direct impacts from local sources.

Data have been provided for 2008–2019 covering 25 stations, although monitoring at one of these was only conducted in 2014. Between 2008 and 2010, sampling around the country was done annually and covered 16–23 stations each year. From 2011 onwards, fish were collected at east and west coast stations on alternate years. For these years, there are data for 14–15 east coast stations (odd years) or 8–9 west coast ones (even years).

All data relate to designated English waters, with the exception of those from a Welsh station in the Bristol Channel as this is a shared water body in which fish are likely to move freely across territorial water.

Typically, 5 or fewer pools of fish were sampled around each station. Each pool comprised 5 fish. Stations at which there was only one sample collected have been excluded from the analysis.

4.22.2 Data structure

Data are for nickel, copper and zinc concentrations in liver. All concentration data are reported in mg/kg wet weight.

The data summaries that were provided consisted of results from individual pool samples taken around the stations. These individual samples were used in the trend and threshold assessments rather than station means for the purposes of the dashboard indicator. This is because the pooled samples are already representative of a mean of 5 fish and this approach allows assessment of trends across the stations and nationally.

Within the datasets of 836 samples per substance, the LoDs were 0–0.23, 0.04 and 3.21mg/kg wet weight for nickel, copper and zinc, respectively. One hundred and forty samples for nickel and one each for copper and zinc were reported below those LoDs; these were assigned values equal to half the LoD.

One sample taken from a station in 2019 was removed as an extreme positive outlier for all three metals.

4.22.3 Exploration of change in chemical concentrations over time

Summary data for nickel, copper and zinc concentrations in dab liver from samples analysed across the period are given in Tables 4.22.1, 4.22.2 and 4.22.3, respectively.

Table 4.22.1 Summary statistics for samples of nickel in dab liver (mg/kg wet weight)[1]

Year	Number of stations	n	Mean	SD	Median	Min	Max	Q1	Q3
2008	23	114	0.14	0.10	0.12	0.00	0.56	0.060	0.19
2009	23	115	0.069	0.074	0.045	0.015	0.53	0.025	0.090
2010	16	79	0.065	0.070	0.050	0.0050	0.38	0.010	0.095
2011	15	74	0.22	0.49	0.10	0.0050	3.4	0.070	0.15
2012	8	40	0.12	0.19	0.075	0.020	1.2	0.050	0.10
2013	15	74	0.16	0.12	0.14	0.0080	0.53	0.080	0.20
2014	9	45	0.25	0.39	0.13	0.0080	2.0	0.070	0.21
2015	14	66	0.28	0.34	0.19	0.030	2.4	0.11	0.29

Year	Number of stations	n	Mean	SD	Median	Min	Max	Q1	Q3
2016	8	37	0.08	0.047	0.070	0.030	0.23	0.040	0.090
2017	15	75	0.11	0.062	0.090	0.030	0.35	0.060	0.15
2018	9	45	0.066	0.029	0.062	0.021	0.16	0.047	0.077
2019	15	70	0.27	0.75	0.10	0.027	6.1	0.063	0.21

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Table 4.22.2 Summary statistics for samples of copper in dab liver (mg/kg wet weight)[1]

Year	Number of stations	n	Mean	SD	Median	Min	Max	Q1	Q3
2008	23	114	5.95	2.15	5.80	1.80	16.0	4.55	6.88
2009	23	115	5.45	2.54	4.90	0.0200	17.0	3.70	7.00
2010	16	79	4.51	1.90	4.20	1.60	12.0	3.15	5.30
2011	15	74	4.10	1.85	3.90	0.340	10.0	2.75	4.90
2012	8	40	5.55	2.78	4.95	2.60	16.0	3.90	6.15
2013	15	74	6.23	2.41	5.99	2.26	14.4	4.49	7.44
2014	9	45	5.85	2.18	5.50	3.10	13.0	4.30	6.40
2015	14	66	6.37	3.50	5.30	2.00	20.0	4.15	7.53
2016	8	37	6.47	3.47	5.30	1.80	14.0	3.20	8.90
2017	15	75	5.47	2.16	5.20	1.80	13.0	3.95	6.55
2018	9	45	5.25	2.01	5.23	1.65	10.6	4.03	6.33

Year	Number of stations	n	Mean	SD	Median	Min	Max	Q1	Q3
2019	15	70	4.98	2.33	4.60	1.59	11.4	3.11	6.58

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

Table 4.22.3 Summary statistics for samples of zinc in dab liver (mg/kg wet weight)[1]

Year	Number of stations	n	Mean	SD	Median	Min	Max	Q1	Q3
2008	23	114	28.0	3.92	27.5	18.0	38.0	25.0	31.0
2009	23	115	23.3	5.12	23.0	1.61	43.0	20.0	25.5
2010	16	79	21.0	3.87	22.0	12.0	29.0	18.0	24.0
2011	15	74	22.7	4.89	23.0	3.50	34.0	21.0	25.8
2012	8	40	25.4	4.56	25.0	18.0	39.0	23.0	27.3
2013	15	74	26.3	3.37	26.4	20.1	34.0	24.0	28.5
2014	9	45	22.3	2.82	22.0	17.0	28.0	20.0	24.0
2015	14	66	26.8	7.17	25.0	18.0	46.0	22.0	28.0
2016	8	37	25.5	5.31	24.0	15.0	38.0	22.0	30.0
2017	15	75	25.6	2.77	25.0	20.0	35.0	24.0	27.0
2018	9	45	23.6	4.45	23.1	15.7	37.0	21.4	25.1
2019	15	70	23.7	3.67	23.8	16.6	33.7	21.4	25.3

[1]n: number of samples analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

To bring the data as close to a normal distribution as possible, the measured concentrations were converted into Ln values for the purpose of assessing trends. Plots of the overall change in nickel, copper and zinc Ln concentrations in dab liver from 2008 to 2019 are shown in Figure 4.22.1.

Figure 4.22.1 Scatterplots of Ln nickel, copper and zinc residues in the liver of dab from marine waters around England. Data shown are for individual samples. The solid black line shows the trend from a generalised additive model as a function of time (diagram courtesy of Cefas)
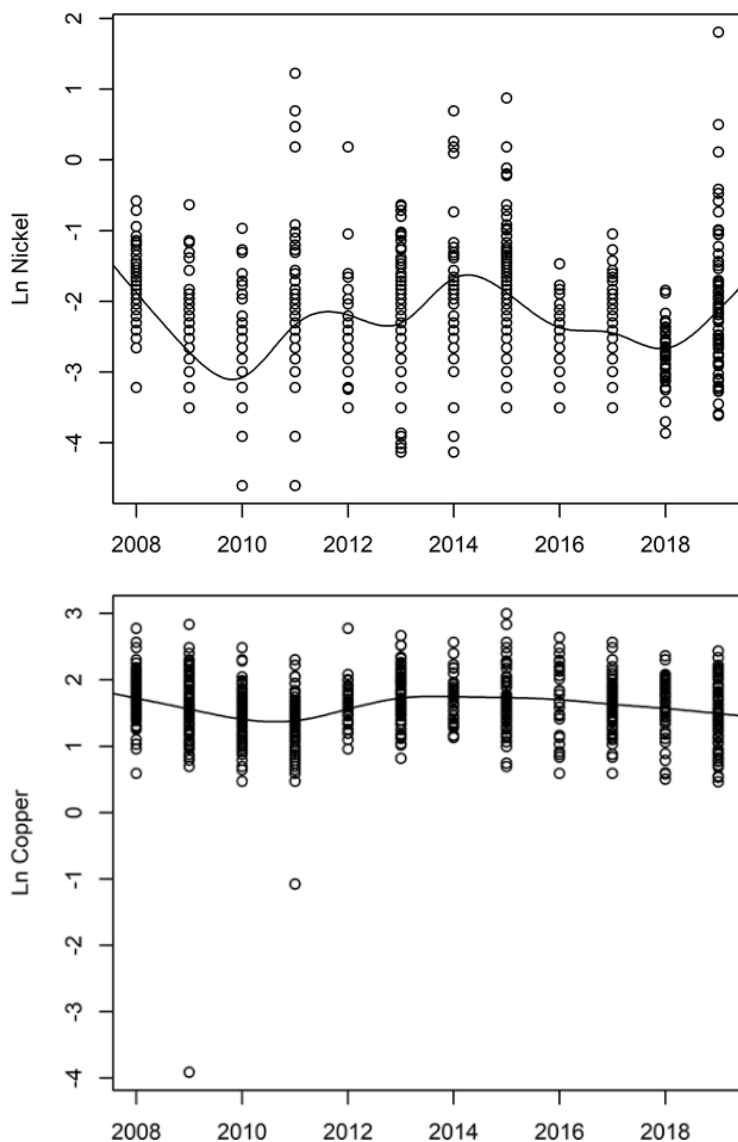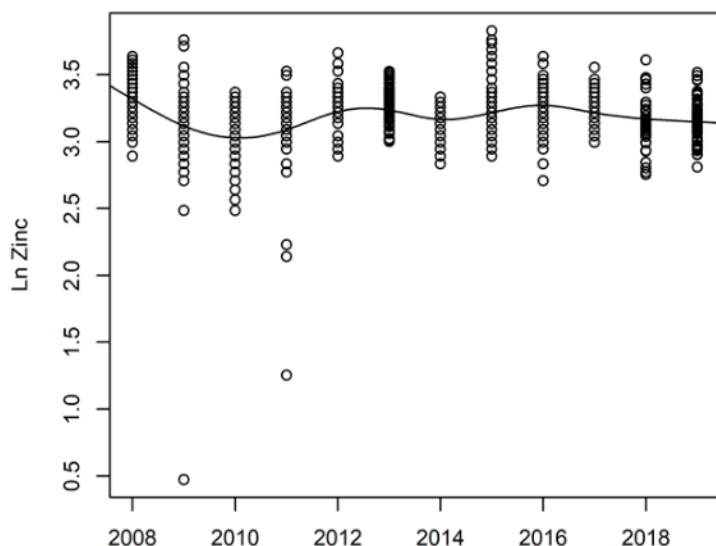
Minimum data requirements for trend assessment are met, though the monitoring regime has altered over time (see Section 4.22.1).

To determine the overall trend for the dashboard, temporal changes in concentrations at individual stations were assessed. This is because it is possible that analysing the results all together may give misleading conclusions for trends and mask intersite variations. Scatterplots of results for samples taken at these stations can be found in Appendix B, Figures B.4.22.1, B.4.22.2 and B.4.22.3 for nickel, copper and zinc, respectively.

A GAM (Wood, 2017) was used for the assessment. The same criteria for interpreting the GAM plots for individual stations as described in Section 4.8.3 was used here. The statistic D for each metal was then calculated as also described in Section 4.8.3.

For nickel, six stations – all on the east and south coast – show upward trends; no downward trends are observed (Figure B.4.22.1). The resulting D value (24%) strongly supports the assignment of increasing trend (↑) and this is used within the dashboard.

Downward trends for copper residues in dab liver are seen at 4 stations; 5 stations show upward trends. There is no obvious geographical split in the results. The D value of 4% puts copper in the no observed change category (↔) and this is used within the dashboard.

In the case of zinc, one station showed a downward trend in the south west and 3 showed upward ones. This results in a D value of 8% and an assignment of no observed change (↔); this is used within the dashboard.

The significance of the limited variation in the concentrations of copper and zinc in dab liver as shown by the mean values in Tables 4.22.2 and 4.22.3 is not yet clear. These values may be a reflection of naturally regulated concentrations in fish as the metals are essential elements.

4.22.4 Thresholds

There are no ecological thresholds either under the OSPAR framework or derived as EQSs. Therefore assessment of the most recent data against thresholds for nickel, copper and zinc is not possible and the dashboard entries indicate that there are no corresponding thresholds defined.

4.23 Pesticides and biocides in freshwater: pesticides

4.23.1 Data source

Data on pesticides in freshwaters are collected as part of the Environment Agency's Watch List surveillance monitoring and Catchment Sensitive Farming (CSF) monitoring. Analysis involves scanning using gas chromatography–mass spectrometry and liquid chromatography–mass spectrometry.

The resultant concentration data from the Watch List monitoring are available from 2016 and those from CSF from 2014, although limited earlier data from 2007 have been collected at fewer sites. Using these data allow the consideration of a far broader suite of pesticides (300+) over time than those historically monitored using traditional quantitative methods.

The assessment of the data used for the dashboard is based on a toxic units approach (Bundschuh et al., 2014). This involves conversion of pesticide concentrations into toxic units (TU) using an ecotoxicological endpoint. This effectively converts concentration data into a measure of the risk posed in a given sample by a given substance.

This TU approach does not rely on consistently evaluating data for the same pesticides each year, as the metric is the total toxic risk posed and not an absolute concentration/quantity of any particular active substance. Thus, the metric is robust to the future introduction or withdrawal of active substances from the market as these will not affect the usage of the indicator, although they may alter the magnitude of the value.

For this report, the ecotoxicological endpoint used was the EC50 for *Daphnia magna*. This is because these data were the most consistently available for each substance in the analysed suite. Ecotoxicological data were obtained from the University of Hertfordshire's Pesticide Properties Database (PPDB) (University of Hertfordshire, 2020). This section matches the assessment reported in Shore et al. (2020). In future, a more-protective approach and long-term view will be taken through the use of chronic rather than acute aquatic endpoints.

The TU for each active substance detected in the water column was calculated as:

$$TU = \frac{\text{Detected concentration (µg/L)}}{\textit{Daphnia magna}\ \text{EC50 (acute)}}$$

Where EC50 values are unbounded (e.g. >25), the lower bound value was used to calculate the toxic unit.

TUs were calculated for all compounds detected in the scans, provided that EC50 data for *Daphnia magna* were available from the PPDB and that their Chemical Abstracts Service (CAS) numbers matched those provided in the PPDB. Compounds that did not match these criteria were excluded.

For each water sample, it is possible to calculate the:

- Summed toxic units for all pesticides detected (TU_{sum}). This measure implies that different active substances may have additive toxic effects on *Daphnia*. Where a site has been sampled repeatedly within a year, the average (median or mean) and maximum TU_{sum} values across all samples for that site can be calculated.
- The maximum TU within each sample (TU_{max}) can also be used as a metric. Use of this metric implies that the toxic effects associated with individual active substances are independent of each other and the highest TU reflects the highest toxic risk. Thus, this measure ignores the potential of additive, synergistic or antagonistic effects associated with the presence of multiple active substances. Where a site has been sampled repeatedly within a year, the average (median or mean) and maximum TU_{max} values across all samples for that site can be calculated.

For the purposes of the current report, TU_{sum} values have been calculated for all identified pesticides with detectable concentrations in each sample.

4.23.2 Data structure

TU_{sum} values are given as annual site summaries for baseline years (2007–2018, but year span varies by site) and for 2019. There are data for 20 sites that are paired by sample date (data available for baseline years and for 2019) and one unpaired site (data for baseline years only). The water column samples had been scanned for up to 328 pesticides for which there were accompanying ecotoxicological endpoint data.

Site median TU_{sum} values are derived from data for between 2 and 444 individual samples per site in the baseline years and between 1 and 11 individual samples per site in 2019. The median number of pesticides detected in individual samples varied between sites and ranged between 2 and 26 in the baseline years and between 1 and 33 in 2019. The maximum number of pesticides detected in any sample was 50 in the baseline years and 47 in 2019.

4.23.3 Exploration of change in chemical concentrations over time

Site median TU_{sum} values are shown in Figure 4.23.1 and data are summarised in Table 4.23.1.

Figure 4.23.1 Toxic unit scores for pesticides in freshwaters from sites monitored in the baseline years (2007–2018) and 2019. Data for individual sites are shown. Upper graph: horizontal lines within plots indicate median values across sites. Lower graph: lines between plots connect values for the same sites from the two time periods and indicate whether TU scores have increased or decreased
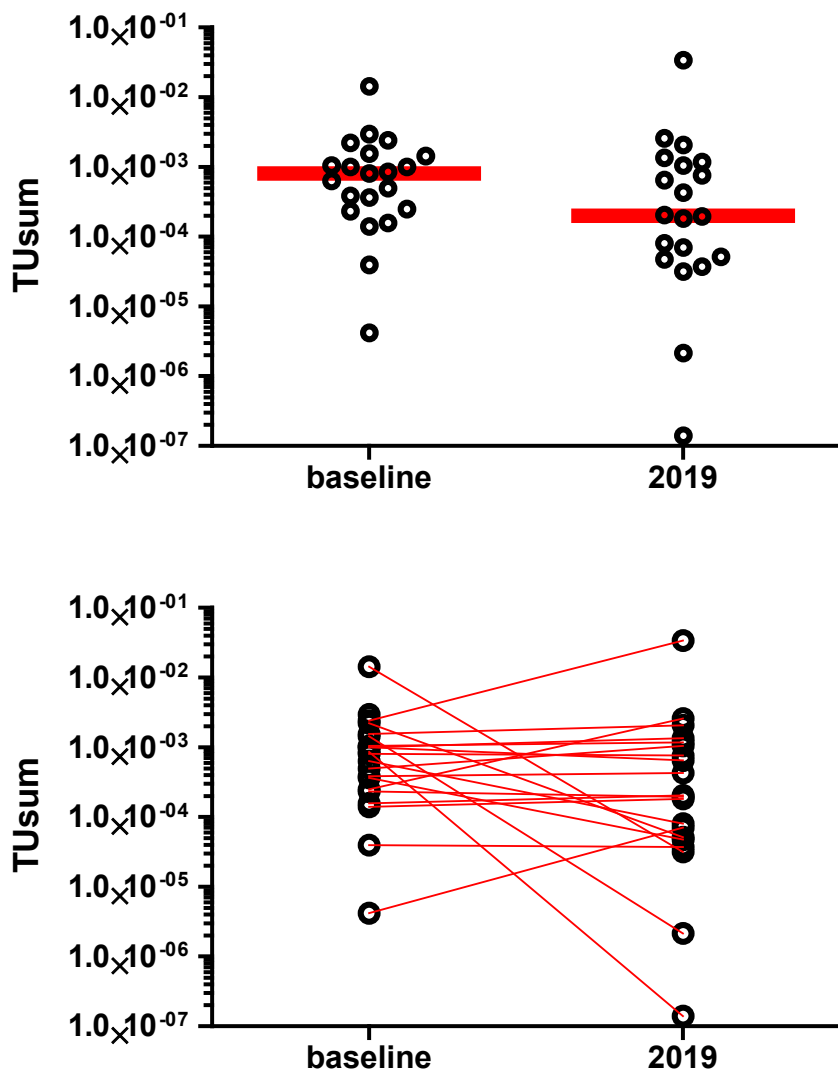
Table 4.23.1 Toxic unit scores for pesticides in freshwaters from sites monitored in the baseline years (2007–2018) and 2019[1]

	Baseline years	2019
Number of sites	21	20
Median n per site	62	4
Median site TU score	0.0008057	0.0002008

	Baseline years	2019
Q1 site TU score	0.00024	4.9×10^{-5}
Q3 site TU score	0.00150	0.00115
Minimum site TU score	0.000004173	1.402×10^{-7}
Maximum site TU score	0.01431	0.03411

[1] TU: toxic unit; n: number of samples; Q1: lower interquartile range value; Q3: upper interquartile range value.

Analysis of the TU_{sum} scores paired by site indicated that there was no statistically significant difference between the baseline years and 2019 (Wilcoxon signed rank test: WE = 4.0, p = 0.96). There was no evidence that differences between sites were maintained between the two time periods as there was no statistically significant association for sites in TU_{sum} scores in the baseline years and 2019 (r_s = 0.13, p = 0.29).

Because the baseline years are compared against data for 2019, the minimum data requirements for reporting a trend assessment have not been met. Therefore, the dashboard entry is left blank.

4.23.4 Thresholds

For this analysis, a Uniform Principle value (UP) of 0.01 TU is taken as an indicative threshold. This value is derived from regulatory information for individual substances that 'the toxicity/exposure ratio for fish and *Daphnia* [should not be] less than 100 for acute exposure [or] less than 10 for long-term exposure' (EC, 2011d). When applied to an overall TU, a UP of 0.01 TUs therefore gives a precautionary assessment of the potential risk posed by pesticides in a sample (for example, as applied by Bighiu et al., 2020; Bundschuh et al., 2014).

One site (4.8% of all sites sampled) had a median TU_{sum} value (based on measurements of four samples) above this value in the baseline years but not in 2019. In 2019, a single (but different) site (5% of all sites sampled) had a median TU_{sum} score that exceeded 0.01, but was based on measurements in only two samples. The TU_{sum} score for this site did not exceed 0.01 in earlier years (median derived from 21 samples). The dashboard entry is based on the number of sites exceeding the threshold in 2019.

4.24 Pesticides and biocides in red kite (*Milvus milvus*): second-generation anticoagulant rodenticides

4.24.1 Data source

Red kites are monitored for their exposure to SGARs because their prey includes rats which are target species subject to control using SGARs. Secondary exposure in red kites is widespread in England and lethal secondary poisoning does occur (Walker et al., 2019).

Data on SGARs in red kite livers have been provided by UKCEH and Fera Science Ltd as part of the PBMS (UKCEH, 2020) and Wildlife Incident Investigation Scheme (WIIS) (Fera, 2020). Necropsy data for kites submitted to the PBMS have been conducted largely by the Institute of Zoology (IoZ), and post-mortem examination of birds submitted to WIIS include birds necropsied by the Animal Plant Health Agency (APHA). The datasets used in this section are drawn from a series of reports (Walker et al., 2017, 2018, 2019).

4.24.2 Data structure

The data consist of measured concentrations of the SGARs brodifacoum, bromadiolone, difenacoum, difethialone, and flocoumafen in the livers of a variable number of red kites found dead each year in England. Summed SGAR concentrations represent the summed concentrations of these 5 compounds.

Data are available for each year between 2015 and 2019 and are reported as ng/g wet weight.

A common LoD of 1.5ng/g wet weight was applied for each individual SGAR, except for difethialone for which an LoD of 3ng/g wet weight was applied. Non-detected concentrations were assigned a zero value when used in the calculation of the summed SGARs.

The data are typically not normally distributed for most years, either as measured concentrations or log10-transformed data.

4.24.3 Exploration of change in chemical concentrations over time

The distribution of data over time is shown in Figure 4.24.1 and summarised in Table 4.24.1.

Figure 4.24.1 Scatterplot of summed SGAR concentrations in the liver of red kites from England. Data shown are for individuals. Horizontal lines within plots indicate annual median values (diagram courtesy of UKCEH)
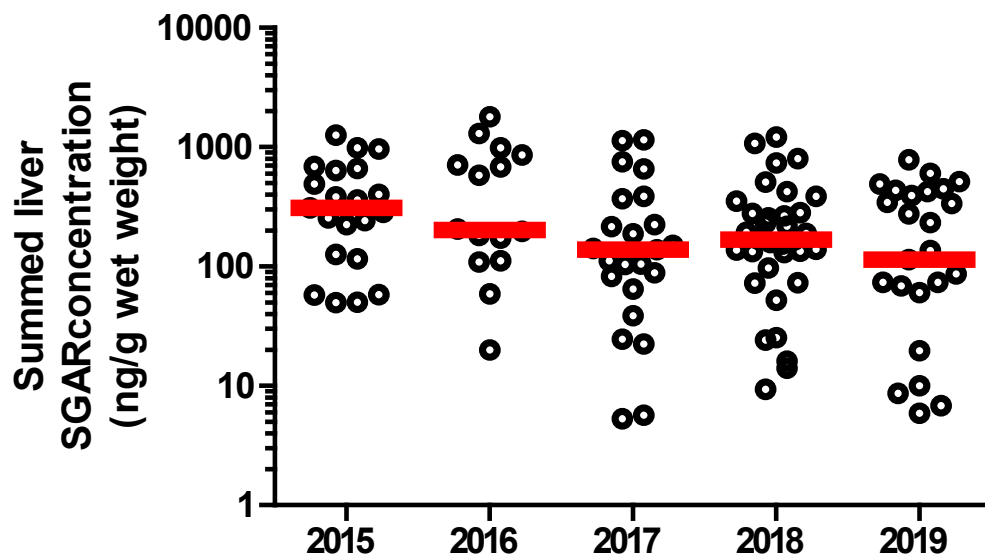
Table 4.24.1 Summary statistics for summed SGAR concentrations in the liver of red kites (ng/g wet weight), and percentage of birds for which it was thought SGAR poisoning had contributed to their death[1]

Year	n	Mean	SD	Median	Min	Max	Q1	Q3	SGAR-poisoned birds (%)
2015	21	411	344	310	50	1266	121	651	33.3
2016	16	500	526	202	0	1800	110	825	43.8
2017	23	269	338	138	5.3	1150	65.0	370	17.4
2018	34	268	291	168	9.4	1218	91.1	302	23.5
2019	27	221	226	114	0	787	10.0	424	7.4

[1]n: number of individuals analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value

There are too few years in the dataset to carry out a regression analysis to test for time trends in the data but a Kruskal–Wallis analysis indicated that there was no statistically significant difference between years in the summed SGAR residues (KW = 7.46; p = 0.114). Because the data do not meet the minimum requirements for trend reporting, the entry on the dashboard relating to trends is blank.

4.24.4 Thresholds

All of the red kites that were analysed were subject to a post-mortem examination conducted by wildlife veterinarians or trained pathology staff at the IoZ, UKCEH, APHA, and Fera Science Ltd. During the necropsy, non-trauma related macroscopic haemorrhaging that was consistent with anticoagulant-rodenticide-induced anticoagulation was noted. Birds were classed as individuals in which SGARs were implicated as a contributory cause of death if such haemorrhaging was present and if anticoagulant rodenticide residues (of any magnitude) were detected in the liver.

There are no statutory threshold values established for SGARs in biota. However, the proportion of red kites in which SGARs are diagnosed as a contributory cause of death is a relevant metric that is considered suitable for use in the dashboard. The proportions affected each year are given in Table 4.24.1. The proportion of kites in which SGARs were implicated as a contributory cause of death in 2019 is used for the dashboard entry (7.4%).

As the sampling strategy for this metric includes submissions to WIIS, which is a targeted collection scheme linked to suspected pesticide incidents including deliberate poisoning incidents, this may skew the data both in terms of overall extent of population contamination and levels of residues in individuals. Consequently, in years where a higher proportion of the samples analysed have come through submissions to WIIS then it would be expected that there might be more incidence of poisoning in the overall sample.

As noted by Walker et al. (2019), 'we do not know how SGAR-induced mortality affects the population dynamics of red kites, as red kite populations in Britain have expanded and continue to do so (Harris et al., 2019)'. It is possible that any current effects do not prevent population growth, but the data have not been analysed against population trends.

4.25 Pesticides and biocides in red fox (*Vulpes vulpes*): second-generation anticoagulant rodenticides

No dashboard entry – insufficient or no comparable data

4.25.1 Data source

The red fox is omnivorous and eats small mammals, including rats and mice living in proximity to humans. They are potentially directly exposed to SGARs through consumption of unprotected bait and secondarily though predation of rodents and other contaminated prey. Foxes therefore provide a measure of exposure across multiple uptake pathways.

Red fox carcasses are submitted each year to the WIIS as part of investigations into suspected poisoning incidents. Investigations in England are conducted by Natural England and involve the collection of such animals. Animals are submitted to the scheme as a suspected poisoning incident, although the suspected active ingredient involved may or may not have been a SGAR.

Foxes were found dead at various rural and urban locations. Livers were collected and analysed from individuals submitted to the WIIS each year. Data on SGARs in red fox livers generated through the WIIS have been provided by Fera Science Ltd.

4.25.2 Data structure

The data consist of measured SGAR concentrations in the livers of a variable number of red foxes found dead each year from 2015 to 2019, excluding 2016, in England. Data are reported as mg/kg wet weight.

Summed SGAR concentrations were calculated as the sum of the concentrations of the 5 SGAR active ingredients used in the UK, namely brodifacoum, bromadiolone, difenacoum, difethialone, and flocoumafen. Zero values were assigned to non-detected concentrations of each individual compound used in the calculation of the summed SGARs.

It should be noted that the exposure data for SGARs in red fox are not currently representative of the overall population exposure. This is because the data are linked to poisoning incidents investigated by WIIS and are therefore pre-selected. This is unlike the red kite data (Section 4.24) which includes non-WIIS generated data.

4.25.3 Exploration of change in chemical concentrations over time

The data available for assessment are shown in Figure 4.25.1 and are summarised in Table 4.25.1. There were too few years of data to allow for time trend analysis. There was no statistically significant difference in the summed SGAR concentrations in liver between years (KW = 1.055, p = 0.79), but sample sizes were small (≤ 8) in most years other than 2019.

The entry in the dashboard is blank and this, in part, reflects the insufficient data.

Figure 4.25.1 Scatterplot of summed SGAR concentrations in the liver of red foxes from England. Data shown are for individuals. Horizontal lines within plots indicate annual median values (diagram courtesy of UKCEH)
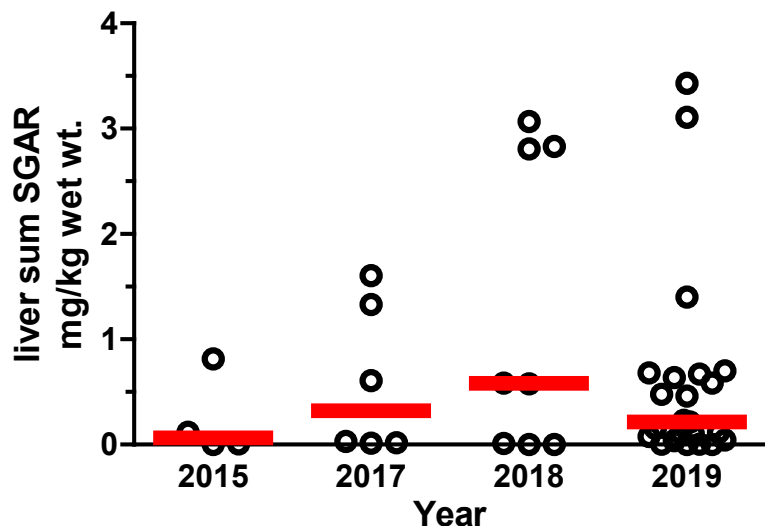
Table 4.25.1 Summary statistics for summed SGAR concentrations in the liver of red foxes (mg/kg wet weight)[1]

Year	n	Mean	SD	Median	Min	Max	Q1	Q3
2015	4	0.235	0.390	0.064	0	0.813	0.002	0.640
2017	6	0.616	0.713	0.320	0.016	1.604	0.018	1.399
2018	8	1.24	1.40	0.582	0	3.070	0.005	2.825
2019	23	0.577	0.918	0.217	0.00006	3.432	0.046	0.673

[1]n: number of individuals analysed; mean: arithmetic mean; SD: standard deviation; min: minimum value; max: maximum value; Q1: lower interquartile range value; Q3: upper interquartile range value.

4.25.4 Thresholds

A threshold has not been established for summed SGAR concentrations in fox livers because there have been limited studies of their effects on foxes. Therefore, no threshold value is proposed for this metric. The blank entry in the dashboard, in part, reflects that there is no value available for comparison.

5. Conclusion

To report our interim H4 indicator, we have updated the work by Shore et al. (2020) to include additional data for some PBTs and heavy metals and expanded our data sources to include blue mussel assessments. Additionally, we have included more years of data for some freshwater and marine datasets to allow the assessment of trends over time.

There is some variability across the different assessments in terms of years assessed, congeners reviewed for PBDEs and PCBs, and in the basis of the thresholds used. Our aim has been to make the assessment comprehensive and consistent as possible using readily available data.

Overall, only a limited number of datasets show statistically significant changes in chemical concentrations over time. This may be a consequence of some chemicals, such as PBT substances, being slow to respond to change. It may also reflect that the data are for a period up to the beginning of the 25-YEP timeline; therefore, some management actions may be in their early stages. Exceedance of thresholds across sites or in individuals is seen for all 3 chemical groups which is not unexpected given the choice of these substances as potential or known substances of concern.

For PBT substances, downward trends in PBDEs, PCBs and PFOS have been observed for marine wildlife in either dab or harbour porpoise. These are particularly evident for PBDEs and the proportion of sites or samples exceeding thresholds is low for this group of substances compared with those for the other PBTs assessed. Exceedance of thresholds is greatest for mercury in the freshwater and marine environments, followed by PCBs in the marine environment. The result for mercury in dab, however, was based on a threshold that could be considered over-precautionary for the tissue examined.

For heavy metals, downward trends have been observed for nickel and zinc in sparrowhawks, though the data are for up to 2014 only. There is an upward trend for nickel in dab at sites in the east and south. The lack of thresholds for some of the data types means it is often not possible to assess risks for this group. Water concentration data for lead, cadmium, nickel, copper, and zinc exceed thresholds at between 1 and 24% of freshwater monitoring sites. This level of risk is also seen for nickel and zinc in estuarine and coastal waters, although the nickel result is only driven by one site. Zinc shows the highest rate of exceedance in both water types.

While the freshwater data for heavy metals show no statistically significant change in concentrations from 2014 to 2019, the freshwater data for metals can be split into two types: those affected by 'abandoned metal mines' and those for sites in 'other' locations. Cadmium and copper show downward trends for the 'other' sites over the assessed time period. No other trends in concentrations over time are seen at either type of site. The elevated levels of most metals in waters affected by abandoned metal mines mean that these sites constitute the majority of those overall exceeding risk thresholds. For nickel, 'other' locations comprise the majority of those overall at risk.

It is not possible to assess trends currently for pesticides and second-generation anticoagulant rodenticides (SGARs). Threshold exceedance is indicated for less than a quarter of sites or individuals considered for a broad range of pesticides in freshwater and SGARs in red kite.

There remain data gaps for the top predators in all compartments for certain substances. Representation of exposure at different trophic levels in the terrestrial environment needs improvement and reintroduction of soil data is a priority so that the entry point to exposure, at least to terrestrial wildlife, becomes known. This will also help contribute to the broader picture of the movement of chemicals in the environment from source to effects in combination with other indicators under the Outcome Indicator Framework.

In filling any data gaps, it will be important to ensure that the data we generate can be reliably gathered to reveal change over time. We will seek to address data gaps for all substances to get a fuller picture across compartments and improve our ability to report exposure trends.

Additional improvements determined from assessing the current data include:

- Re-evaluation of the trends assessments for mussels and the handling of PCB data with varying LoDs.
- Assessing exposure to the common and ubiquitous congener PCB118 to allow more comparability across compartments and improve our understanding of change across different environments and up the food chain.
- Altering the pesticides assessment to consider long-term exposure and the associated risk.
- Determining a definition for negligible or background concentrations and considering emerging substances so that substances can be removed or introduced to the dashboard over time.

Our dashboard indicator has been independently reviewed and received support as an exposure indicator. Some of the points above also reflect comments and recommendations from the reviewers (HSAC, 2020; ECP, 2020) and those from Shore and Walker (2020) from their initial trial of the indicator.

Finally, as strongly recommended through the review process, we need to look at possibilities for reporting on effects within the indicator so that we can reflect the direct environmental impact of chemicals over time. Work is ongoing linking effects from pesticides to populations of macroinvertebrates. We are considering how to refine that information and broaden our understanding of effects across other compartments and organisms.

We will continue to explore options for all points raised through work on the indicator and its review as part of its continuing development.

Acknowledgements

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Appendix A Derivation of an empirical water threshold for perfluorooctanesulfonic acid

As part of the EU EQS derivation process, QSs are derived for different protection goals. These may cover the protection of water- or sediment-dwelling communities, human health, or predators from secondary poisoning. Typically, the lowest QS indicating the most-sensitive protection goal is proposed as an environmental quality standard (EQS).

For PFOS, the EQS is a quality standard based on the protection of human health. However, to consider the risk to freshwater wildlife from PBT substances it is appropriate to use secondary poisoning quality standards (QS$_{sec pois}$). These standards help protect wildlife from the effects of eating prey contaminated by PBT substances.

A QS$_{sec pois}$ for PFOS of 33µg/kg wet weight (EC, 2011b) has been derived through the EU EQS derivation process.

In the EQS dossier (EC, 2011b), equivalent water values to the QS$_{sec pois}$ have been derived: 0.002µg/l in freshwater and 0.00047µg/l in marine waters. These values are considered tentative because of uncertainty around some of the data relating to biomagnification used for the conversion.

The Environment Agency has derived an approach to translate biota standards for PFOS into water concentration thresholds that equate to the same levels of protection. This is based on observations from co-located water and fish sampling data. PFOS is the only PBT substance for which a relationship between concentrations in water and fish was observed. This analysis applies to freshwater only.

Data from 65 freshwater locations sampled from 2015 to 2019 were used to derive the relationship, though not every site was monitored each year. Water samples were taken monthly. Fish were collected on one occasion per year at the same site, but not during the same visit as for water monitoring. The amount of fish data varied between sites: 40 sites had 1 year of data (between 3–5 samples), 8 sites 2 years (6–10 samples), 7 sites 3 years (11–15 samples), 9 sites 4 years (16–20 samples), and 1 site 5 years (21–25 samples) worth of data.

The data were plotted in two graphs (Figure A.1) and analysed using linear regression. The upper graph shown in Figure A.1 is based on overall site means – 1 point per site – and the lower is based on annual site means – 1 point per year per site. Additionally, these data were assessed using a Spearman's rank correlation analysis. The overall statistics from these analyses are given in Tables A.1 and A.2.

Figure A.1 Overall site means (upper) and annual sites means (lower) of PFOS concentrations in freshwater and biota (fish) with linear regression analysis. The diagonal blue lines shows the regression line with the grey shading either side giving the 95% confidence bands. Horizontal dashed lines mark the EQS (human health protection goal) and the QS$_{sec\ pois}$ of 33µg/kg wet weight. The vertical dotted lines indicate the water concentration corresponding to these thresholds based on the regression line.
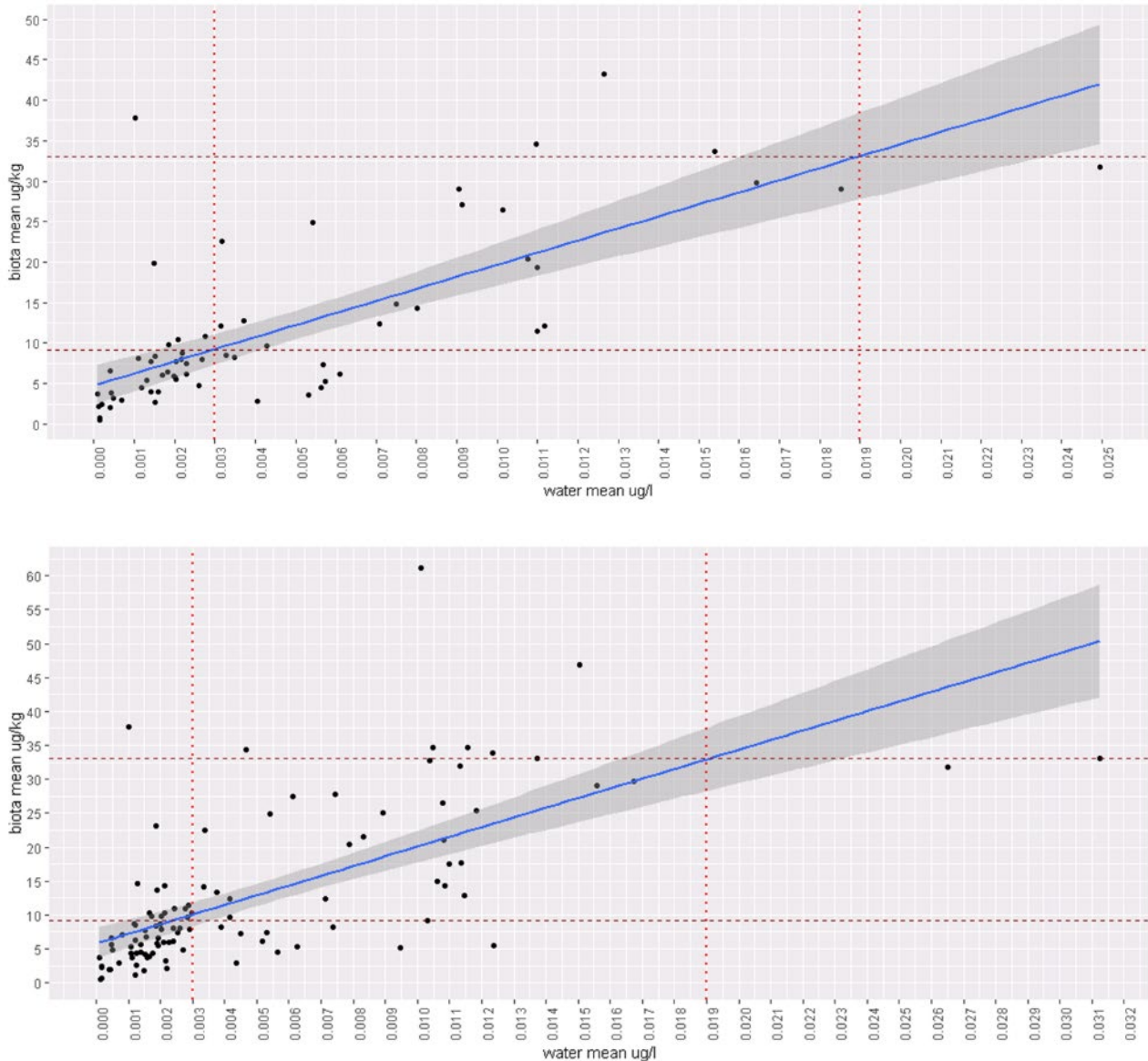
Using the lines of best fit, an equivalent water concentration value of 0.019µg/l was derived for the QS$_{sec\ pois}$. There is slightly more variation in the lower plot (Figure A.1), but the results are very similar.

The linear models are statistically significant as both p-values are below the significance level of 5%. From the Spearman's rank correlation analyses, the correlation coefficients (r_s) show a strong relationship and the corresponding p-values indicate this is unlikely to be by chance (Tables A.1 and A.2).

Table A.1 Statistics from the linear regression model and Spearman's rank analysis of the overall sites means of PFOS concentrations in freshwater and biota (fish)

Test	Statistic	Result
Linear regression model	t-stat	8.449
Linear regression model	p-value	6.611×10^{-12}
Linear regression model	Model p-value	5.872×10^{-15}
Linear regression model	Multiple R-squared	0.5312
Spearman's rank correlation analysis	r_s	0.7243
Spearman's rank correlation analysis	p-value	9.124×10^{-12}

Table A.2 Statistics from the linear regression model and Spearman's rank correlation analysis of the annual sites means of PFOS concentrations in freshwater and biota (fish)

Test	Statistic	Result
Linear regression model	t-stat	9.154
Linear regression model	p-value	6.061×10^{-15}
Linear regression model	Model p-value	5.617×10^{-15}
Linear regression model	Multiple R-squared	0.4486
Spearman's rank correlation analysis	r_s	0.7114
Spearman's rank correlation analysis	p-value	$<2.2 \times 10^{-16}$

The water concentration value derived using this approach that is equivalent to the EQS is used as part of assessing classification in England, alongside the biota value. This is in agreement with the Water Framework Directive (Standards and Classification) Directions 2015 (UK Government, 2015), which allows for monitoring alternative biota or matrices

providing the equivalent level of protection is achieved. However the EQS is based on the protection of human health and therefore not appropriate for use under H4.

A thresholds task and finish Group for the H4 indicator considered the use of the empirical water threshold corresponding to the $QS_{sec\ pois}$. In the sample data, there were fewer samples with high concentrations of PFOS in both matrices, which caused the confidence band width to increase on the regression plots as the values increased. Using the annual data as an example, the $QS_{sec\ pois}$ of 33µg/kg wet weight gives a 95% confidence range of approximately 0.0162 to 0.0232µg/l around the empirical value of 0.019µg/l. Predicting the biota concentration from the empirical water standard value gives a 95% confidence range of between approximately 28 to 37.5µg/kg.

The potential inclusion of the uncertainty from the 95% confidence band in the threshold was discussed because of the slight difference in results seen using the two different media. However use of the value derived from the line of best fit – or midline – was maintained as a more even approach for the following reasons:

- To reduce the chance of false positives and negatives
- Because precaution is already incorporated into the $QS_{sec\ pois}$, and
- Because it is consistent with the current compliance approach.

A value of 0.019µg/l is selected for use within the H4 indicator as a threshold for the protection of freshwater wildlife from the effects of secondary poisoning. This value is more stringent than that derived – through the EU EQS derivation process – to protect the freshwater community from the direct toxic effects of PFOS (EC, 2011b). It is therefore considered protective of wildlife from both routes of exposure.

Appendix B Marine stations map and plots

The Clean Safe Seas Environmental Monitoring Programme (CSEMP) stations considered within this report are mapped in Figure B.1.

The map can be used alongside the scatterplots within this section to gain a spatial understanding of where the different trends are observed. The scatterplot figures are numbered here according to which main section the data are discussed within the report. For example, mercury in marine fish (dab) is covered under Section 4.8, therefore the corresponding diagrams can be found in Figure B.4.8.1. Stations numbered up to and including 494 are on the east coast; those numbered from 534 upwards are on the west coast.

Figure B.1 Map showing the Clean Safe Seas Environmental Monitoring Programme monitoring stations around the English coast
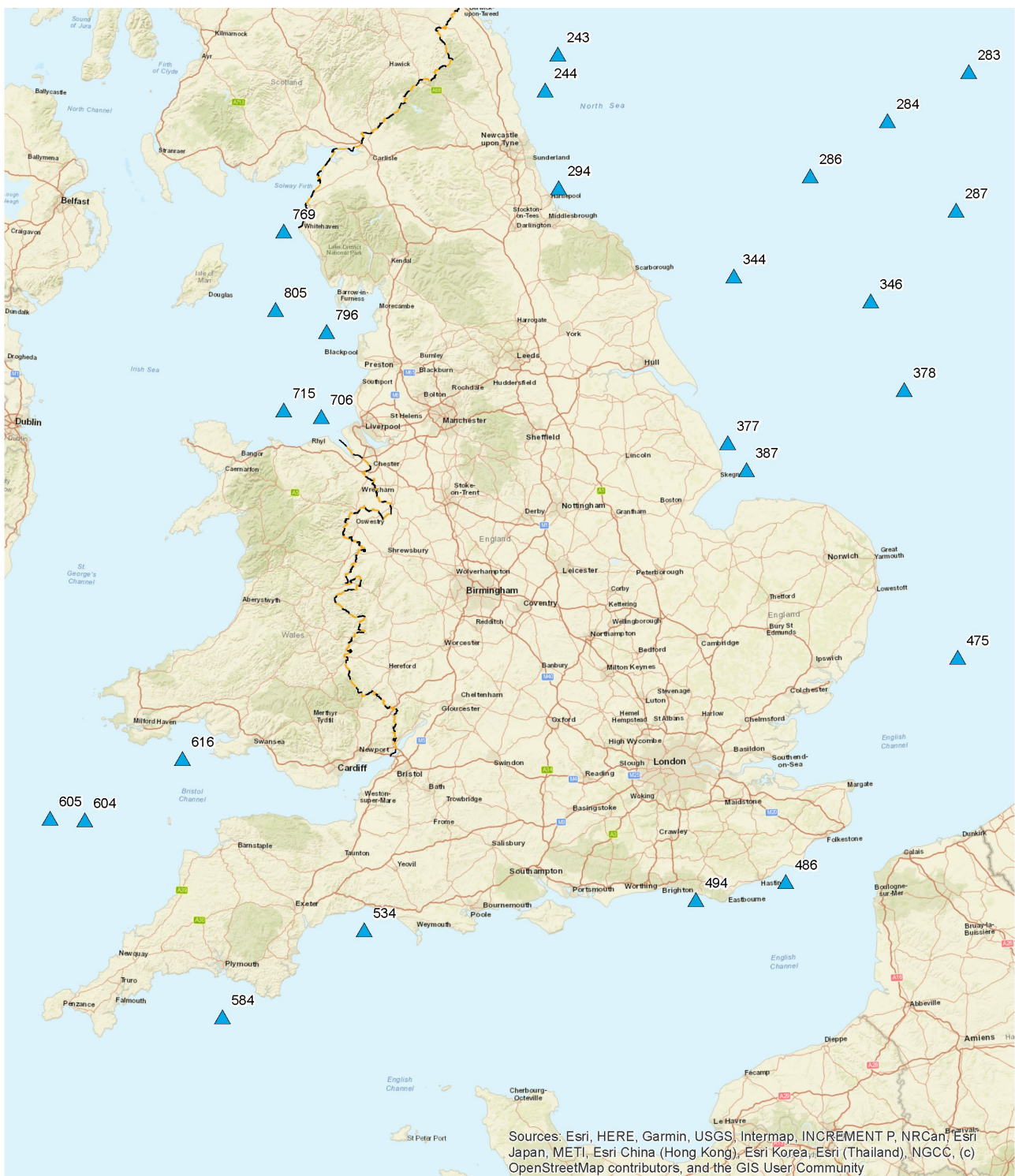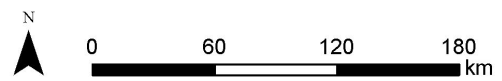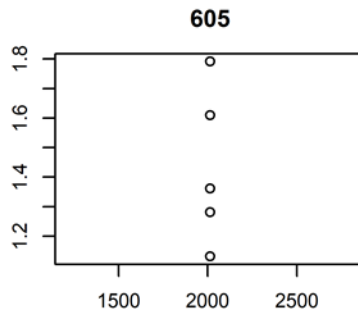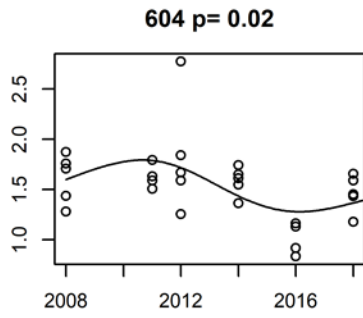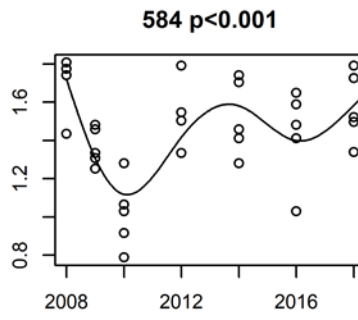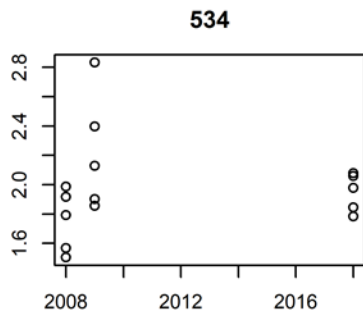
Figure B.4.8.1 Scatterplot of Ln mercury residues in the muscle of dab at individual stations around England. The solid black line shows trends from a GAM as a function of time (diagrams courtesy of Cefas)
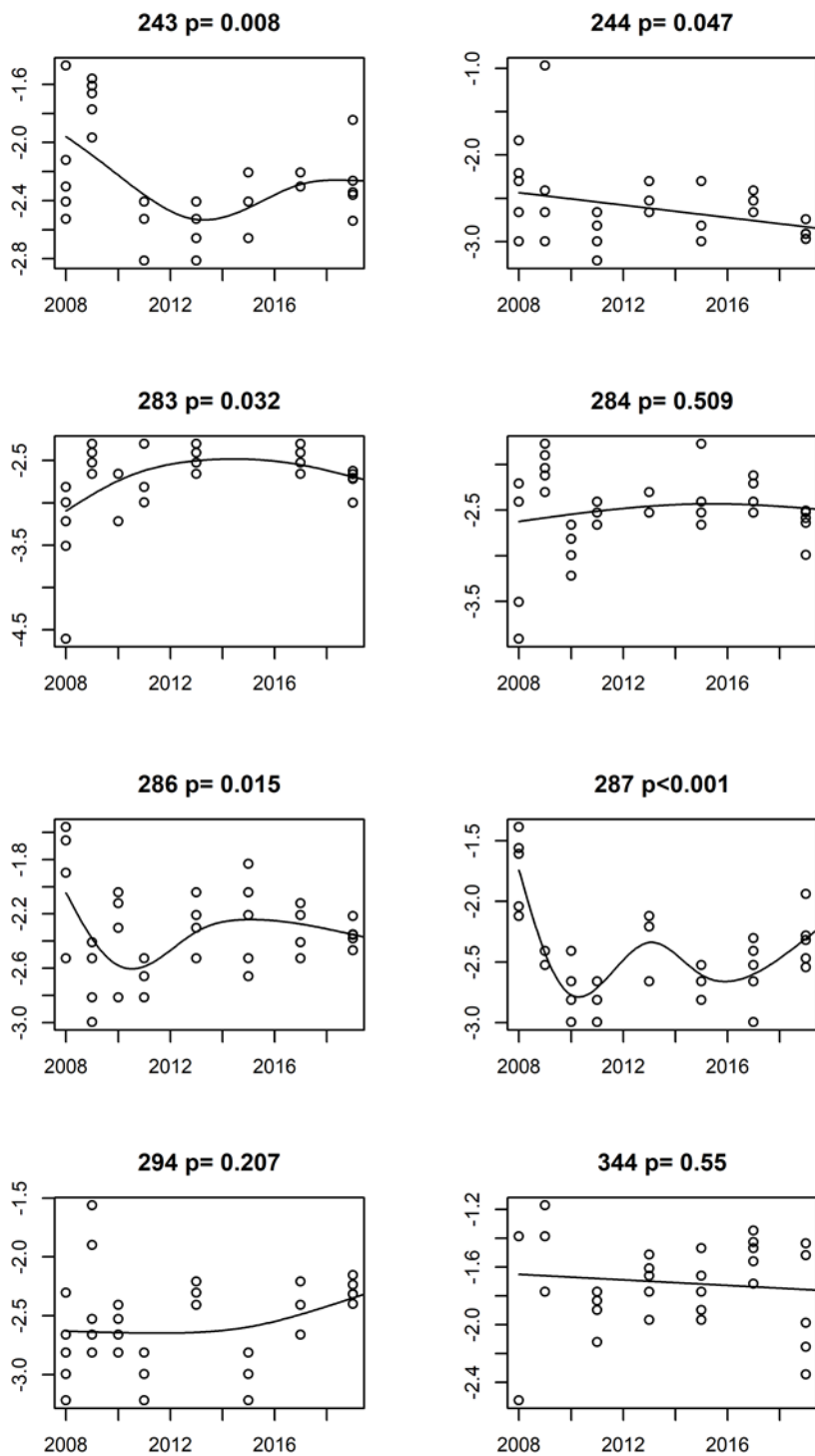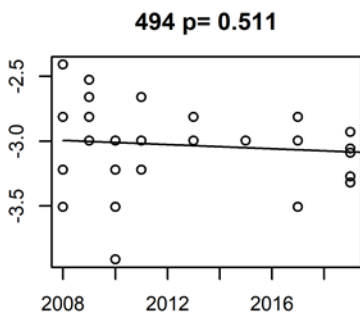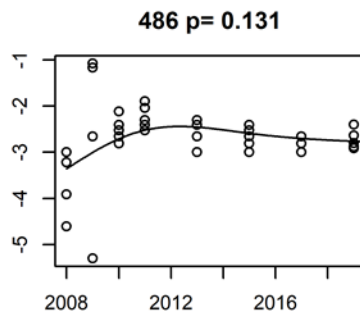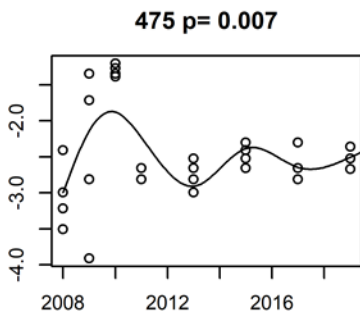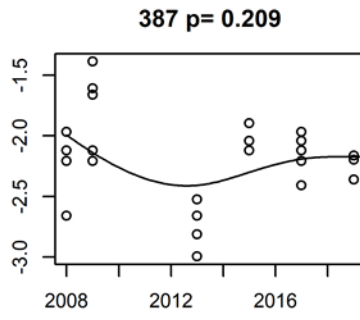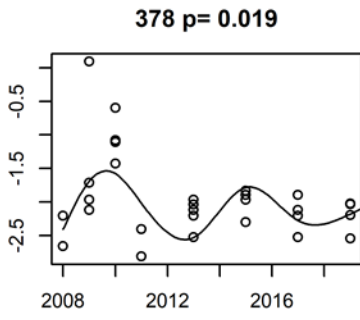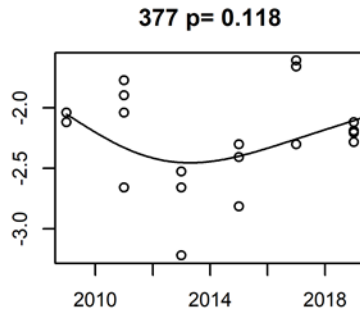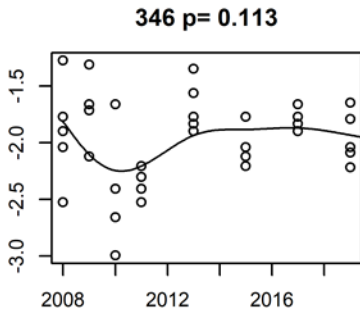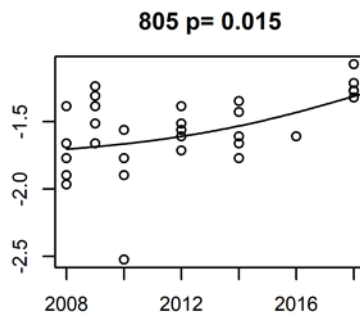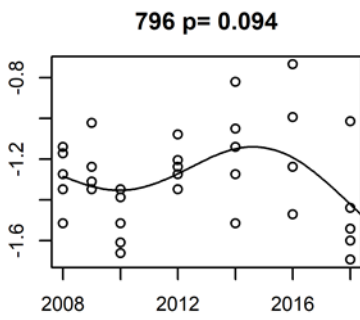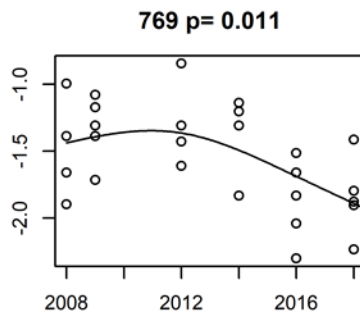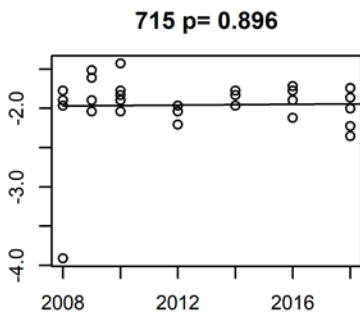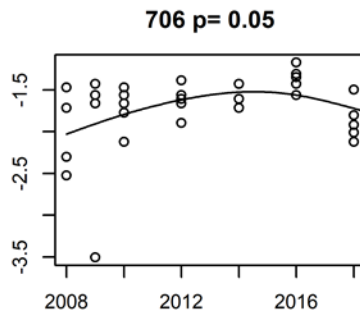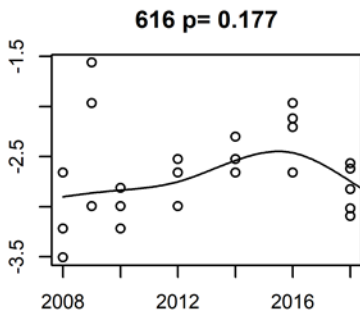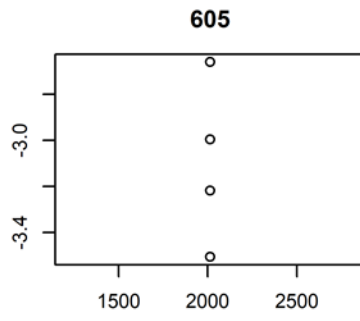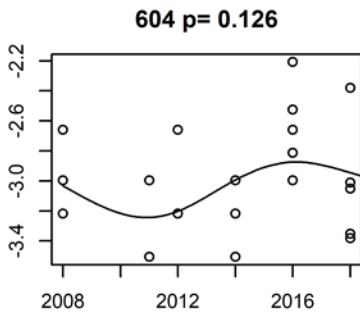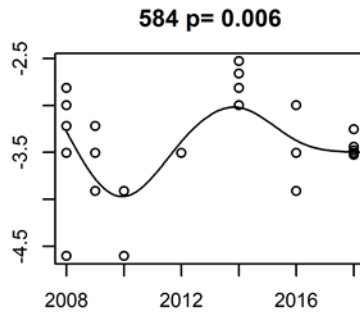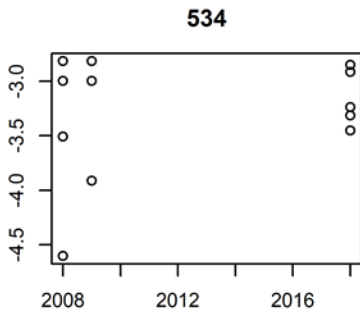
Figure B.4.9.1 Scatterplots of Ln SUM 11PBDE residues in the liver of dab at individual stations around England. The solid black line shows trends from a GAM as a function of time (diagrams courtesy of Cefas)
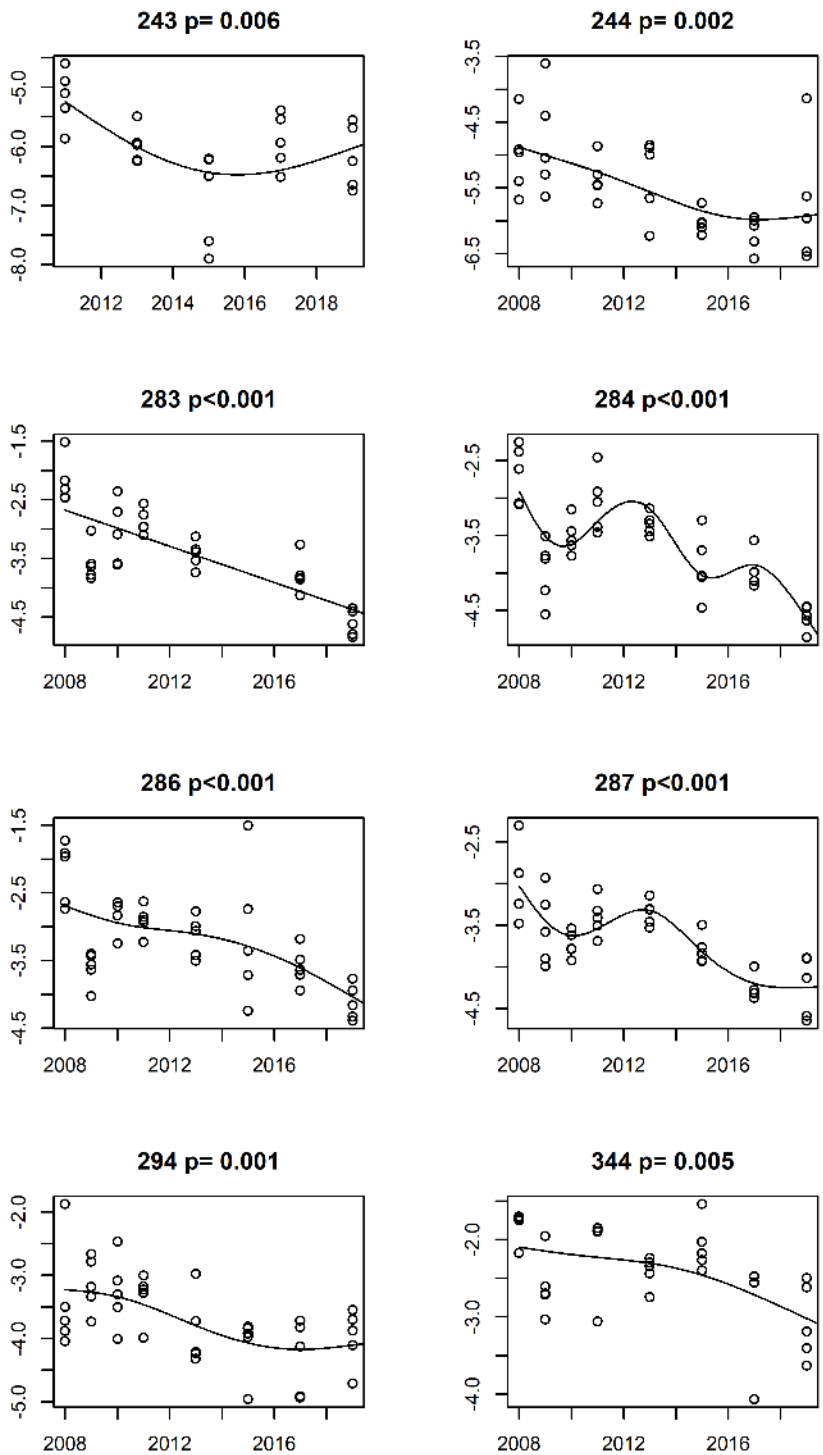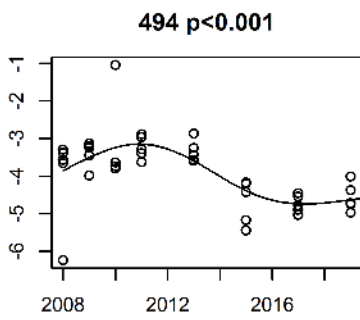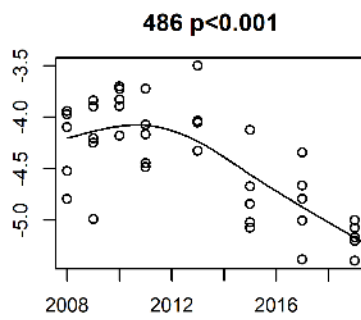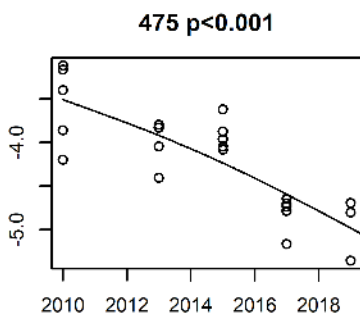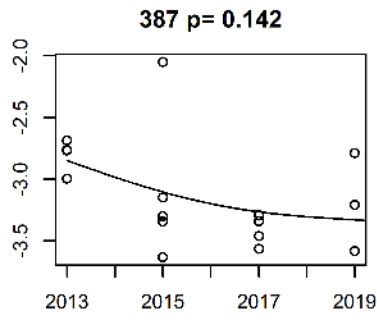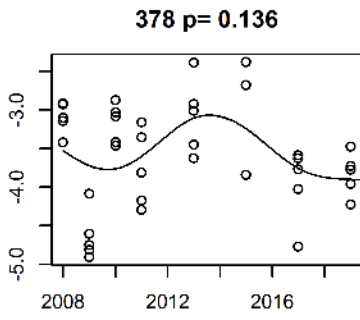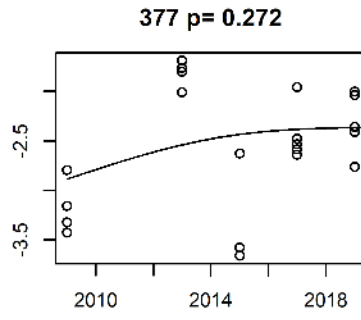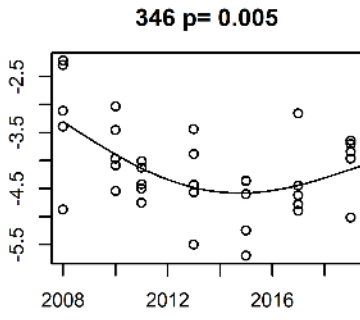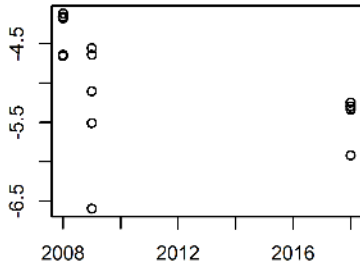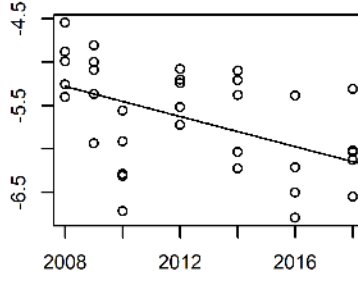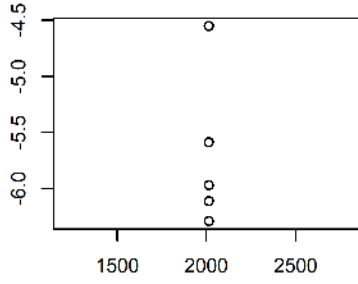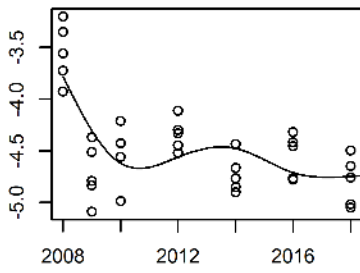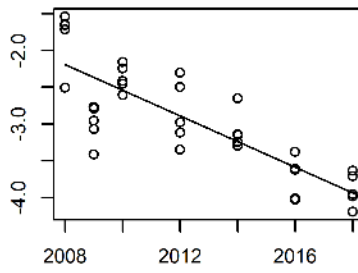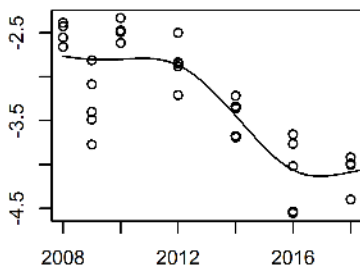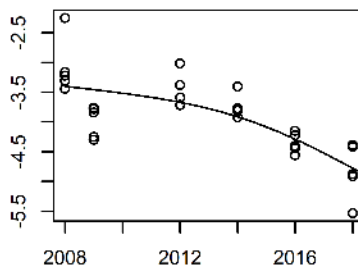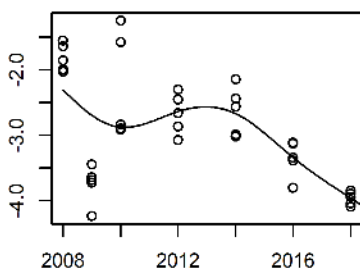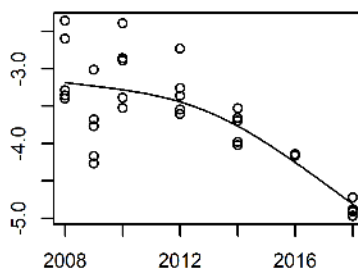
Figure B.4.9.2 Scatterplots of Ln SUM 25PCB residues in the liver of dab at individual stations around England. The solid black line shows trends from a GAM as a function of time (diagrams courtesy of Cefas)
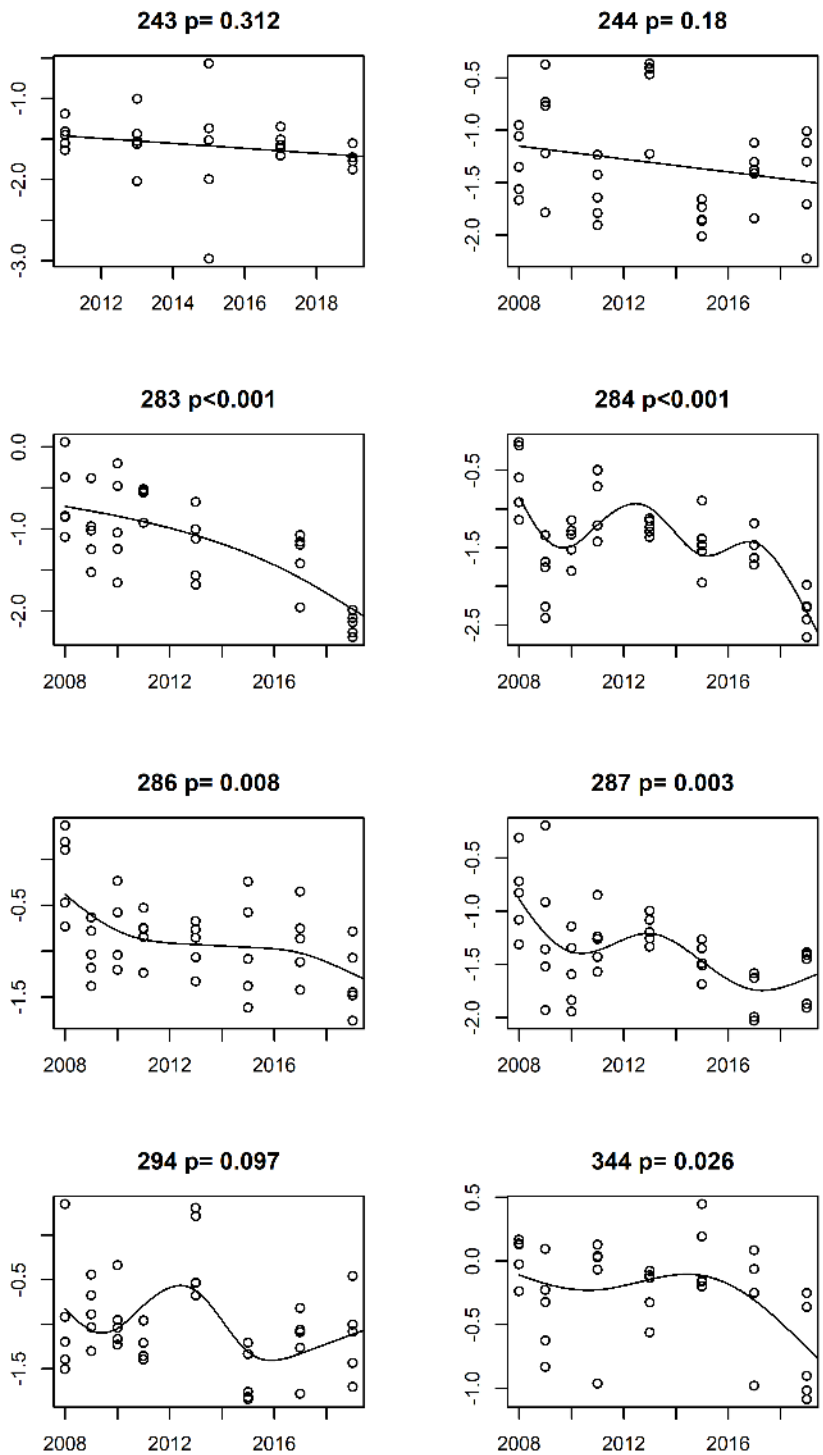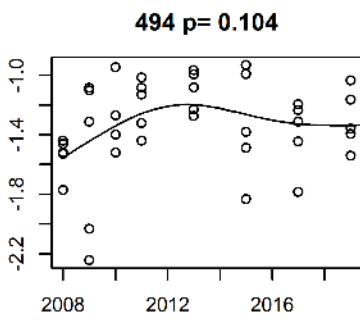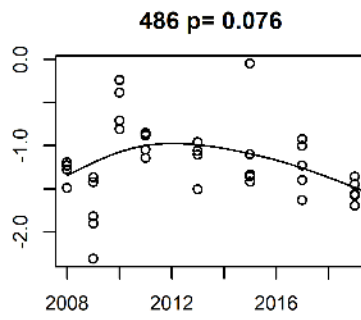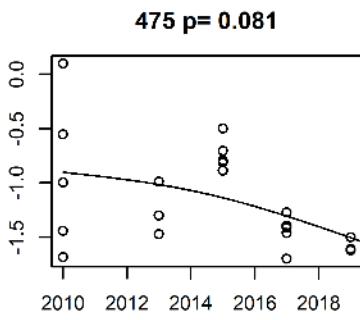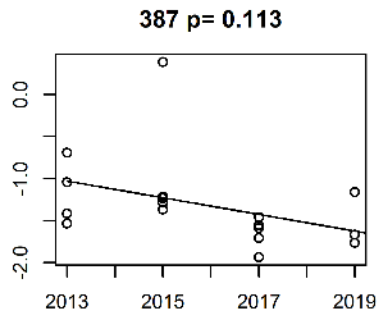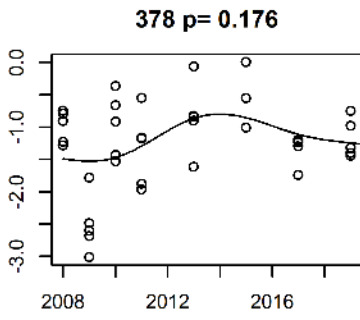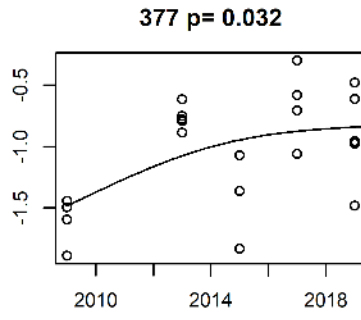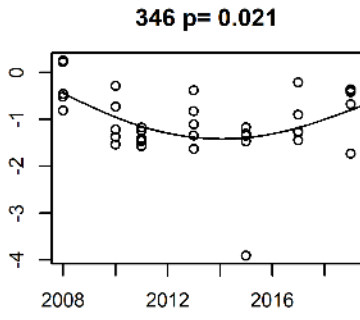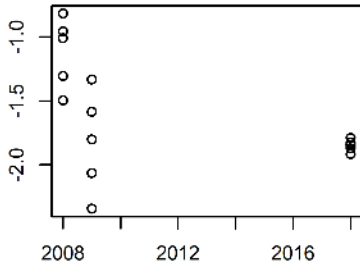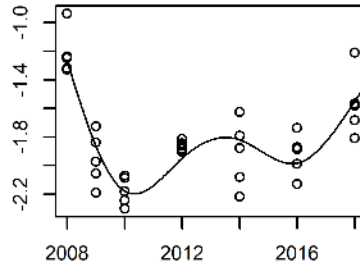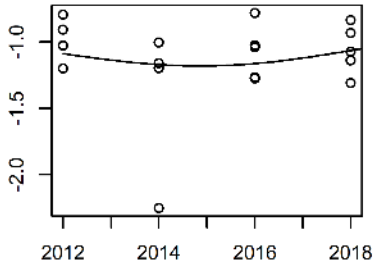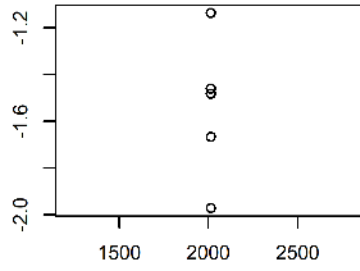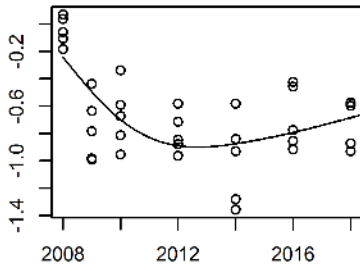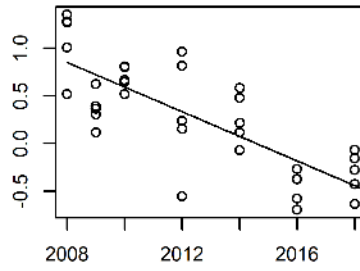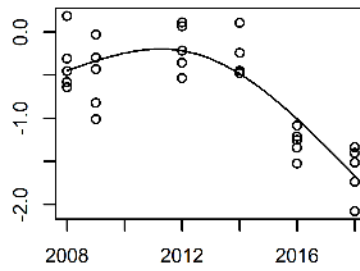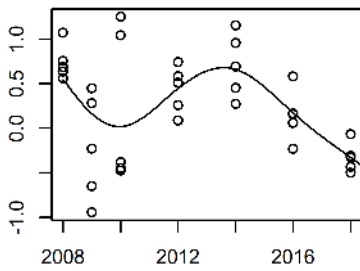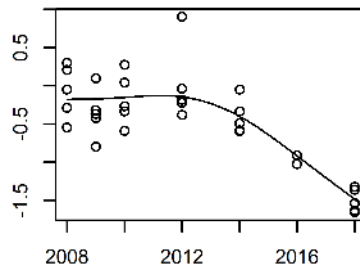
Figure B.4.9.3 Scatterplots of PFOS residues in the liver of dab (µg/kg wet weight) at individual stations around England. The first two diagrams represent east coast station results for 2015 and 2019, the last diagram shows west coast results for 2014 and 2018; means are represented by horizontal lines – black for 2014 or 2015 and grey for 2018 or 2019. Result for stations 287, 378, 387, 486, 707 and 805 were shown to be statistically significant (diagrams courtesy of Cefas)
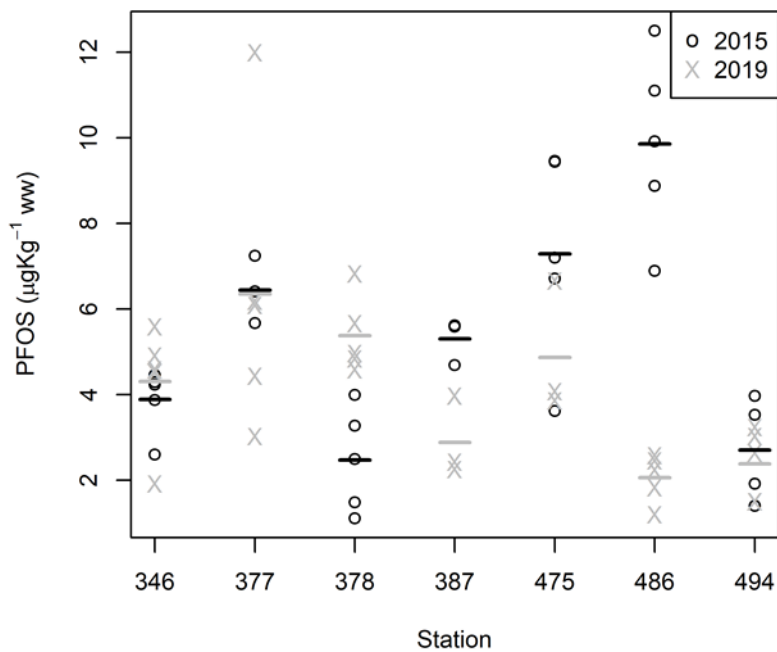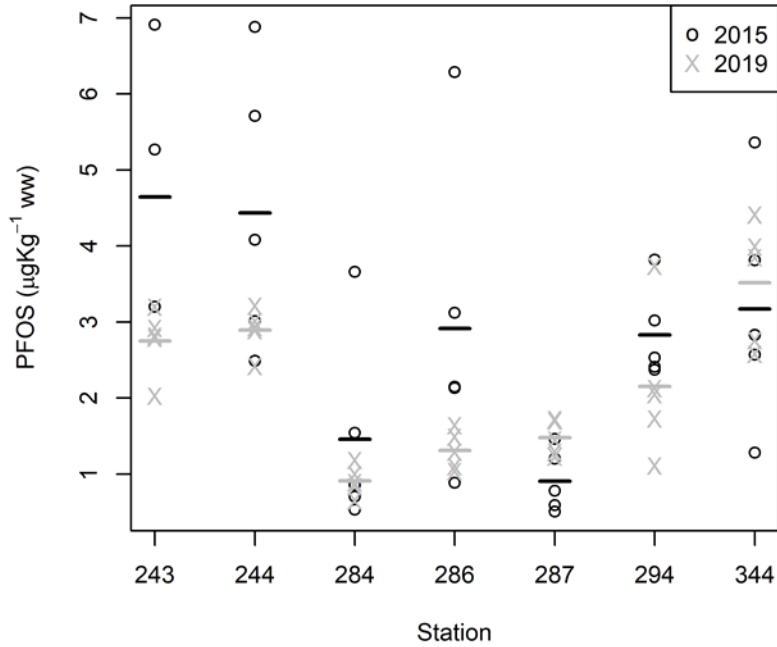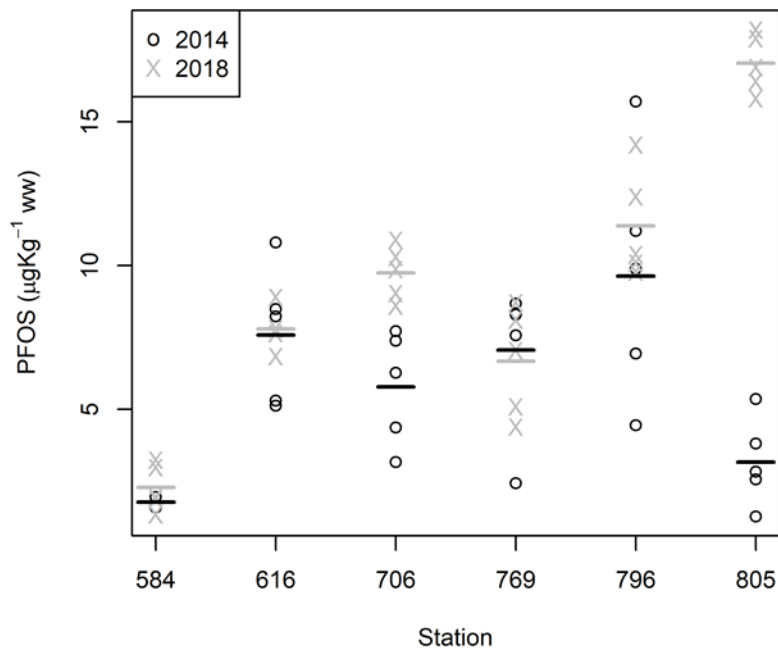
Figure B.4.21.1 Scatterplots of Ln lead residues in the liver of dab at individual stations around England. The solid black line shows trends from a GAM as a function of time (diagrams courtesy of Cefas)

616 p= 0.047

706 p= 0.028

715 p= 0.134

769 p= 0.004

796 p= 0.96
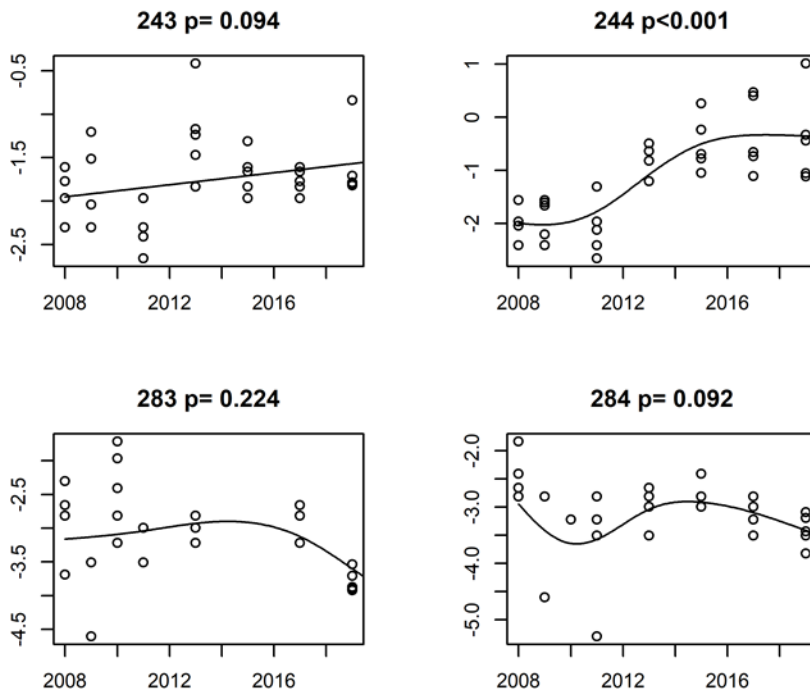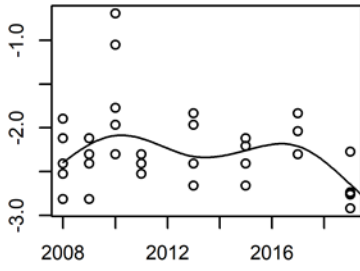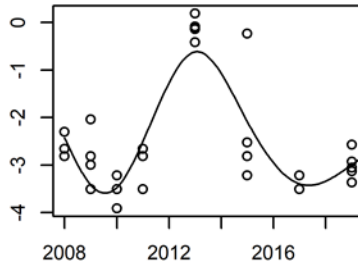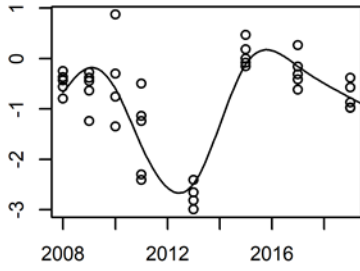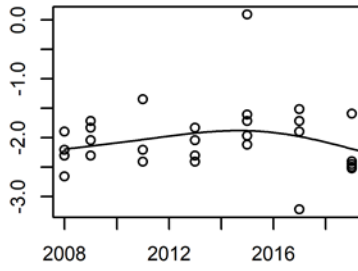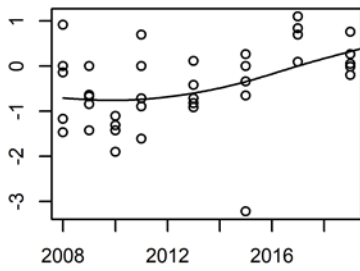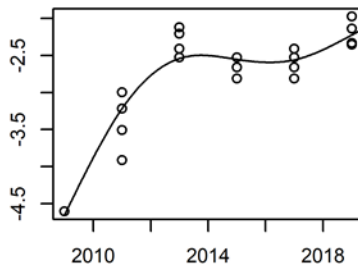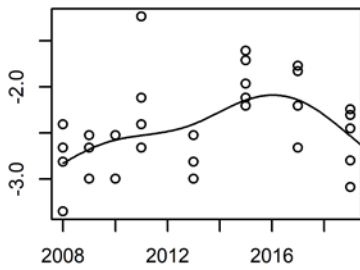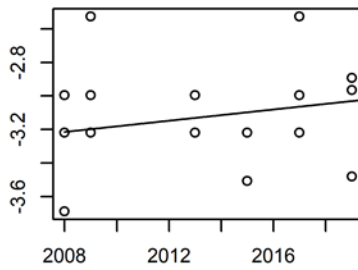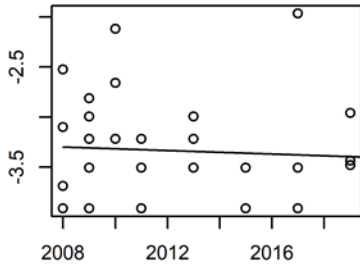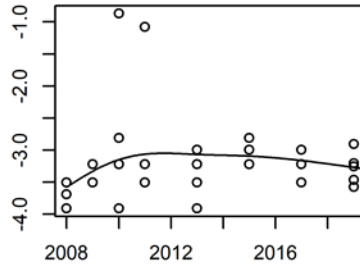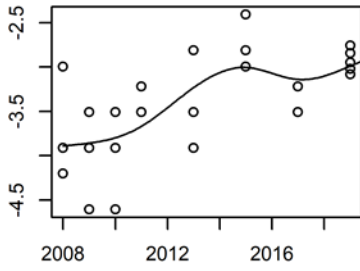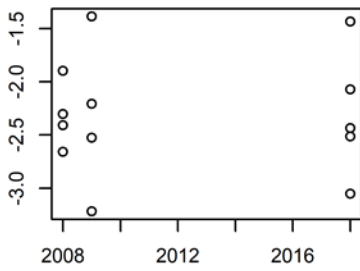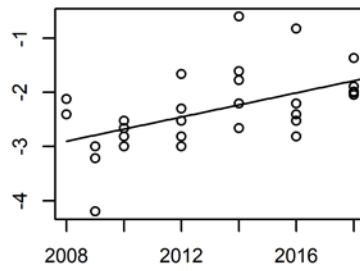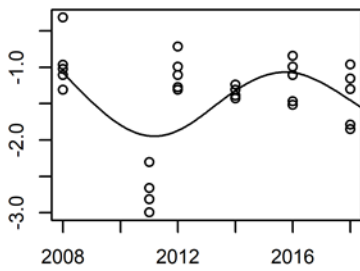
805 p= 0.035
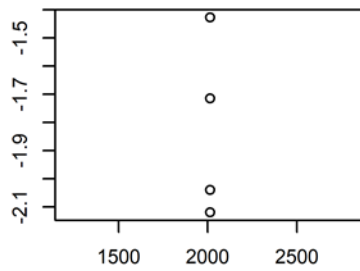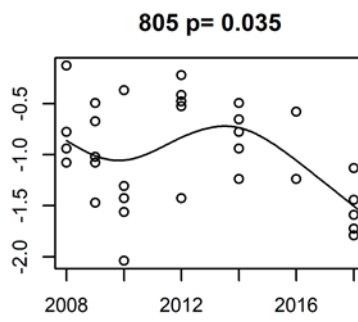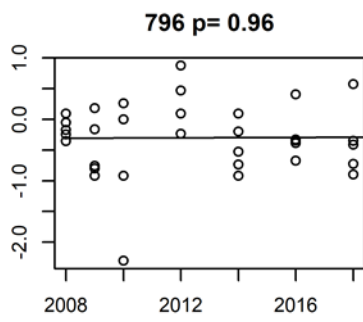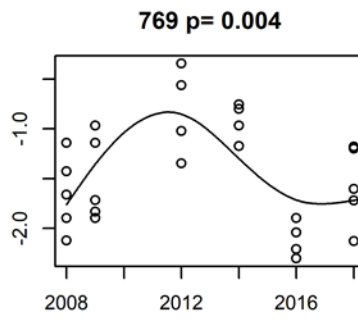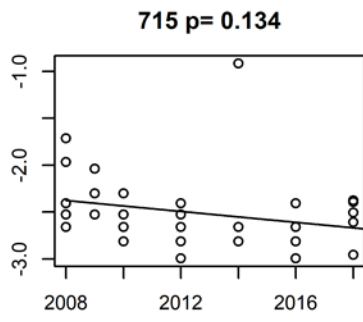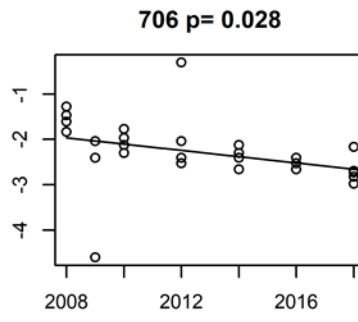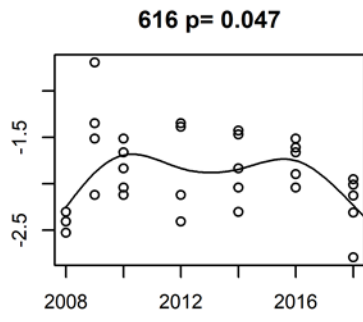
Figure B.4.21.2 Scatterplots of Ln cadmium residues in the liver of dab at individual stations around England. The solid black line shows trends from a GAM as a function of time (diagrams courtesy of Cefas)
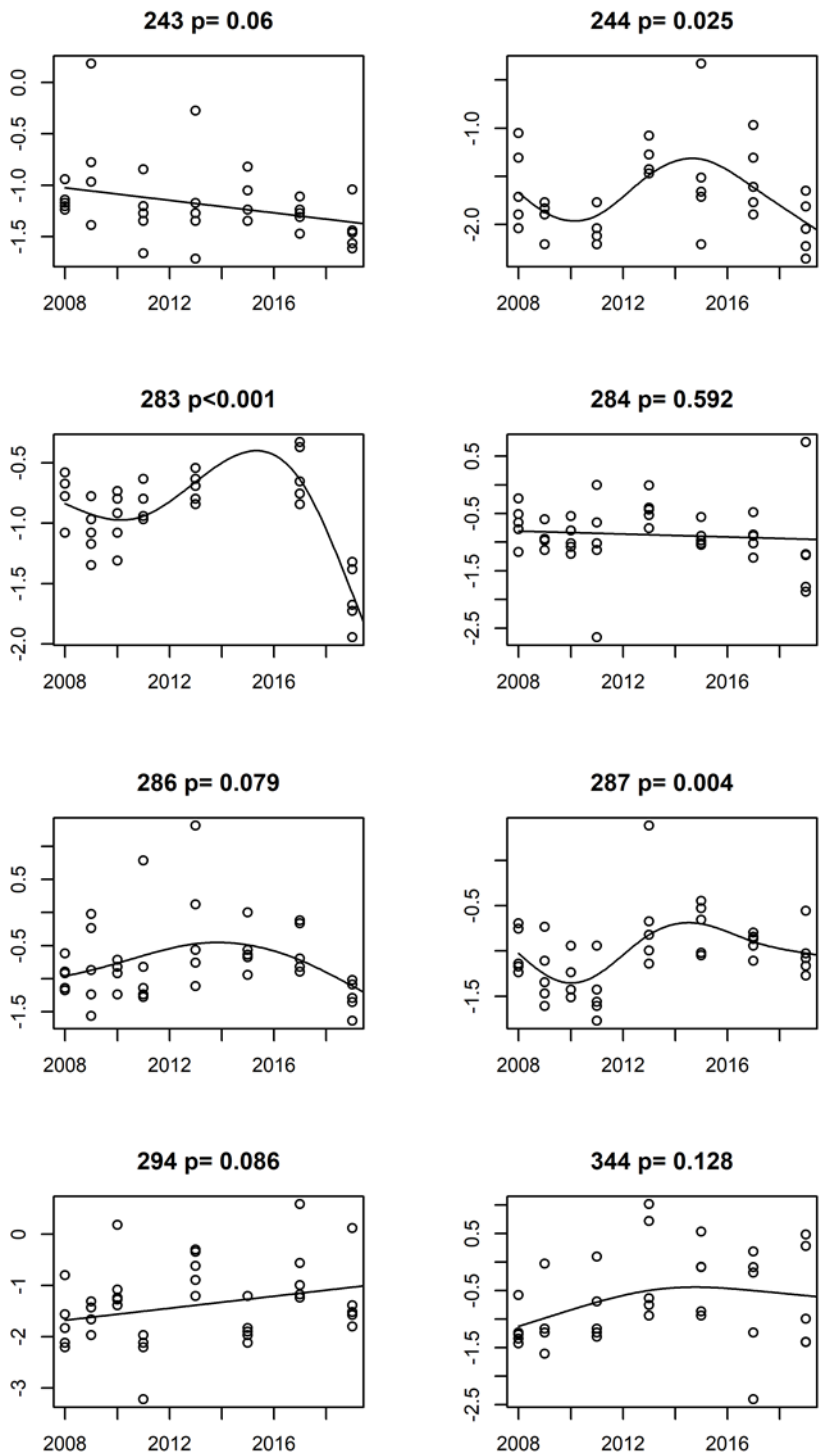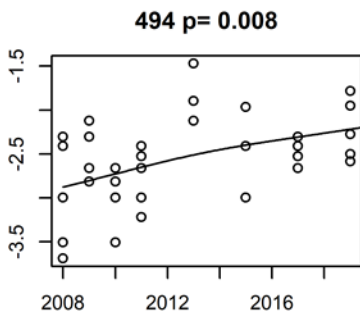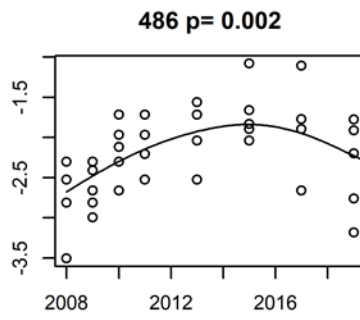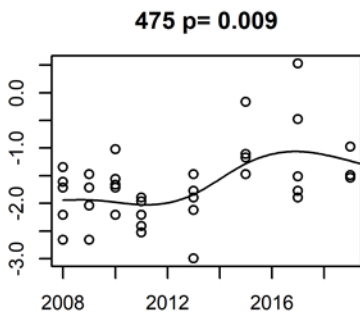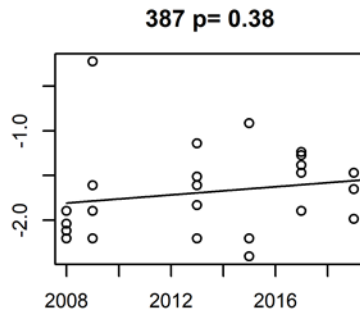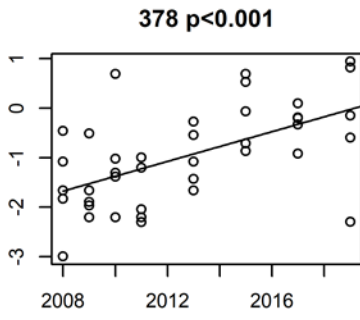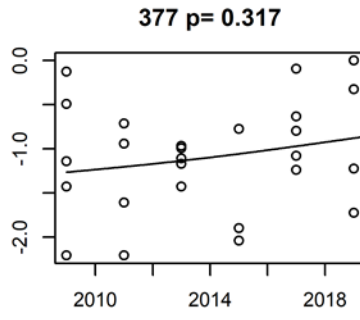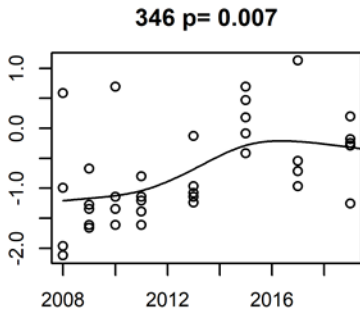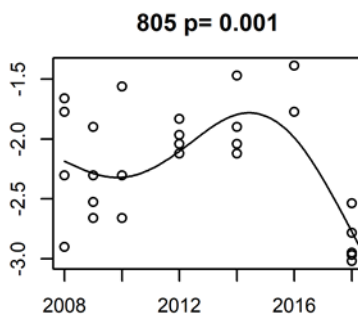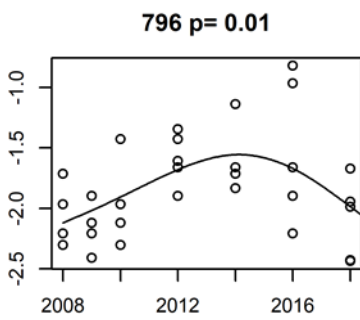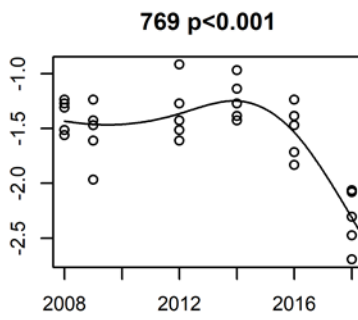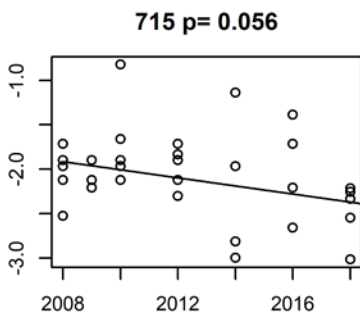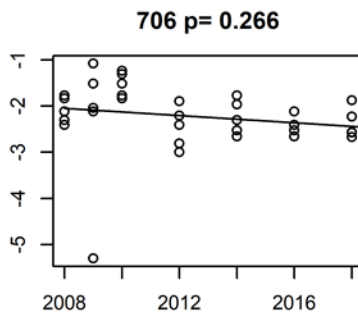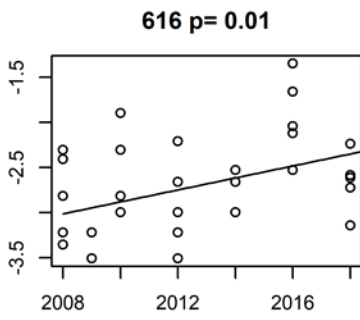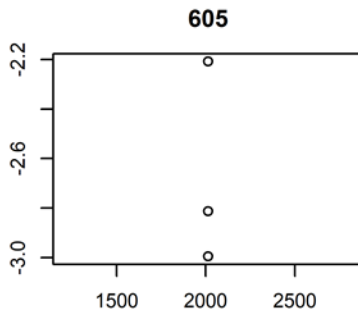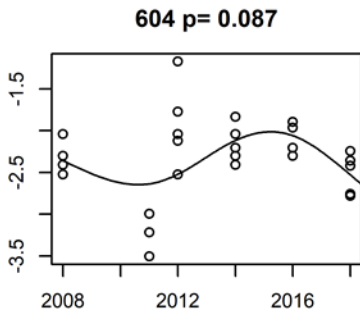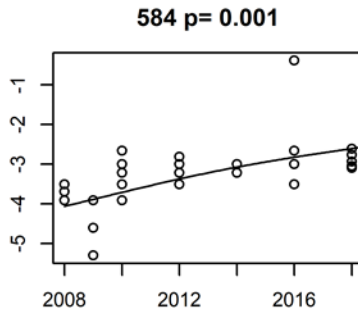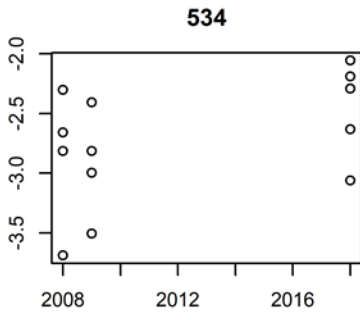
Figure B.4.22.1 Scatterplots of Ln nickel residues in the liver of dab at individual stations around England. The solid black line shows trends from a GAM as a function of time (diagrams courtesy of Cefas)

346 p= 0.005

377 p<0.001

378 p= 0.002

387 p= 0.014

475 p= 0.09

486 p= 0.364
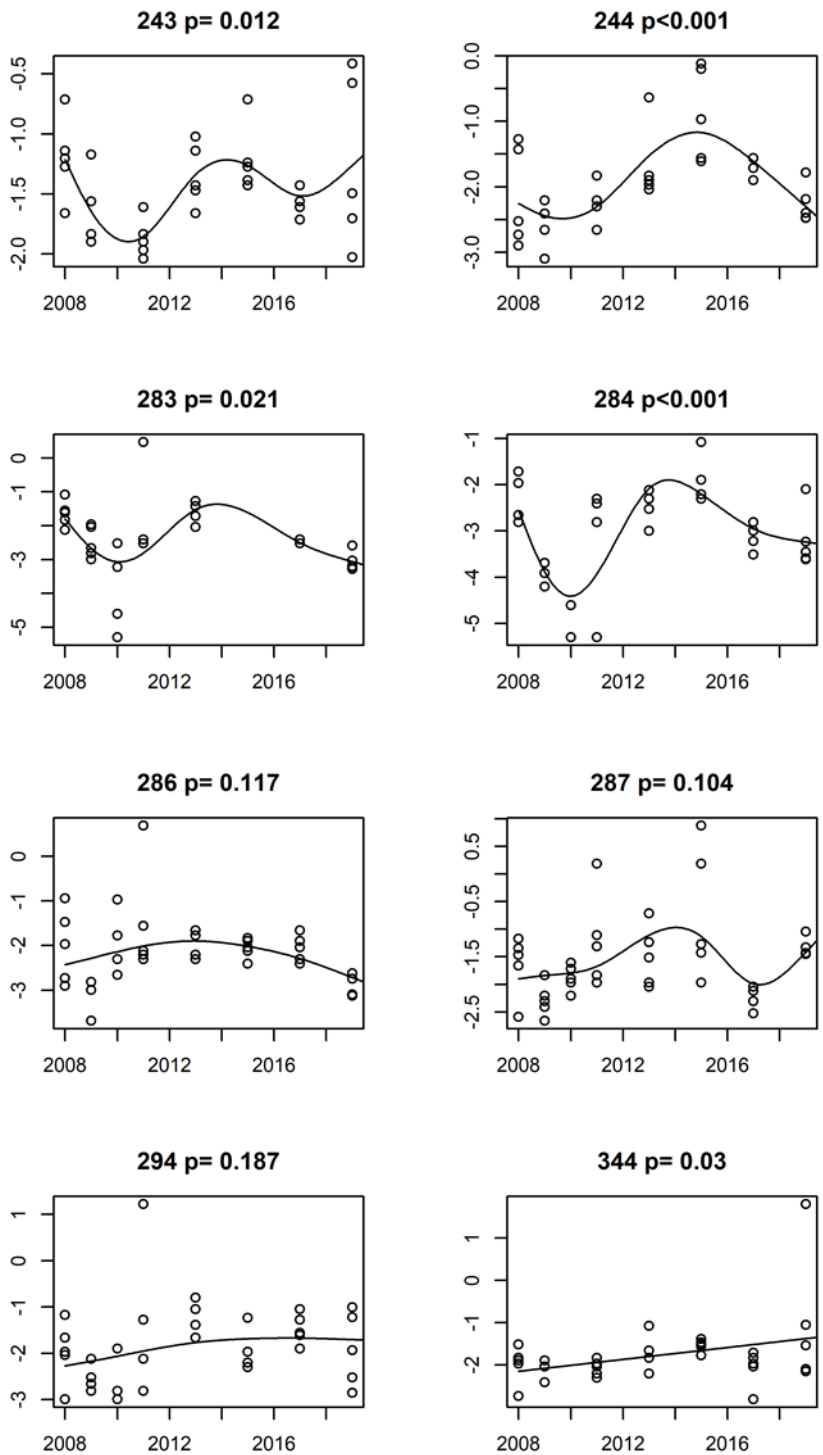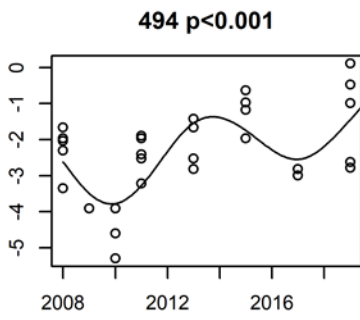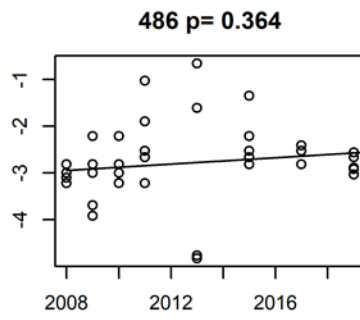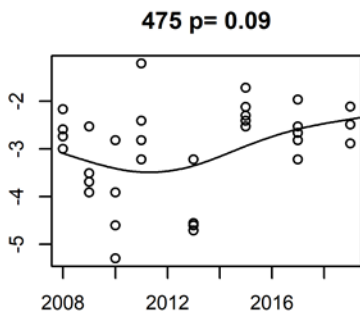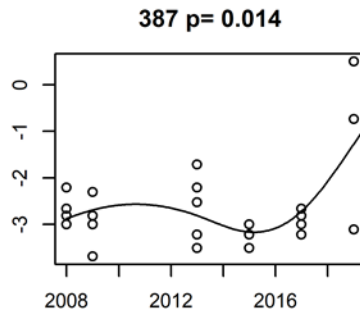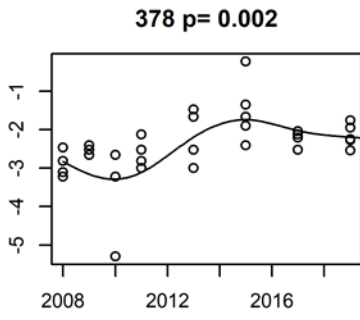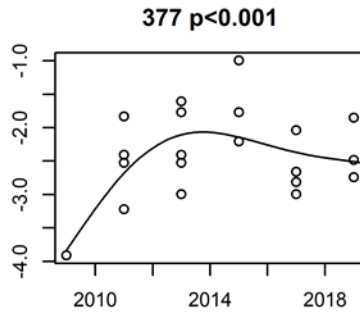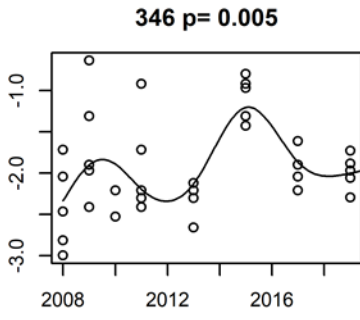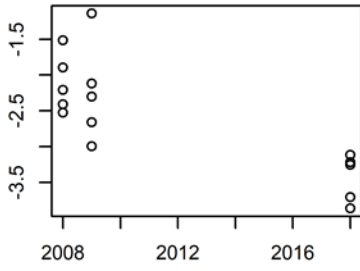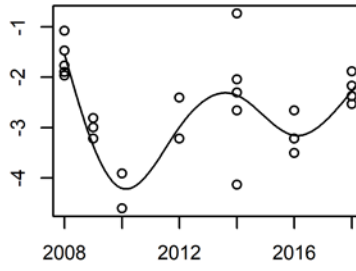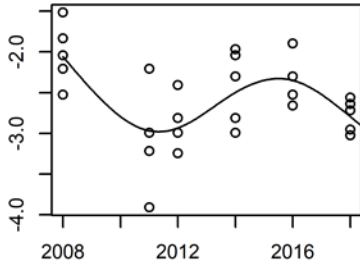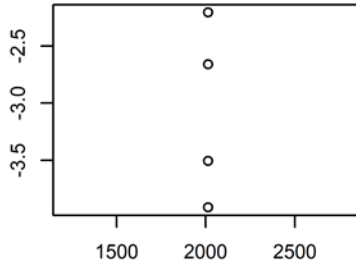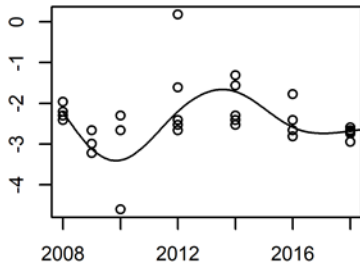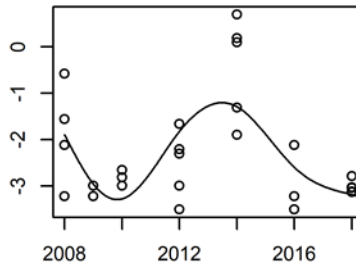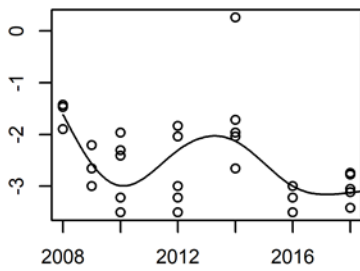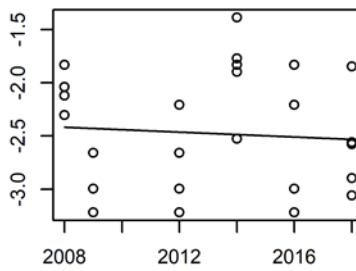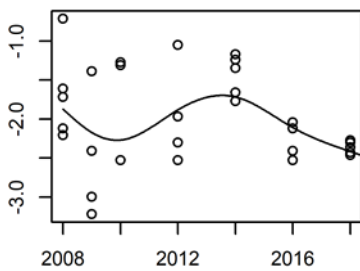
494 p<0.001
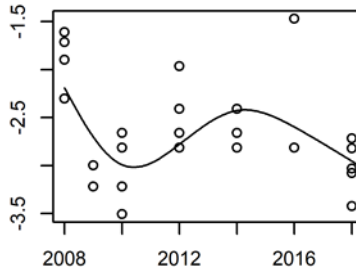
Figure B.4.22.2 Scatterplots of Ln copper residues in the liver of dab at individual stations around England. The solid black line shows trends from a GAM as a function of time (diagrams courtesy of Cefas)

346 p= 0.005   377 p= 0.021

378 p<0.001   387 p<0.001

475 p= 0.001   486 p<0.001
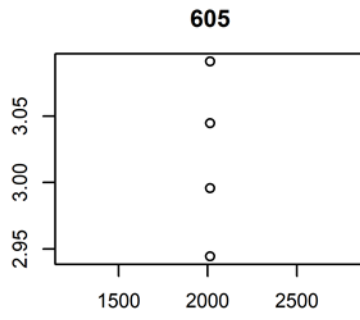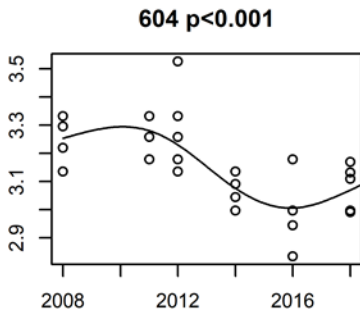
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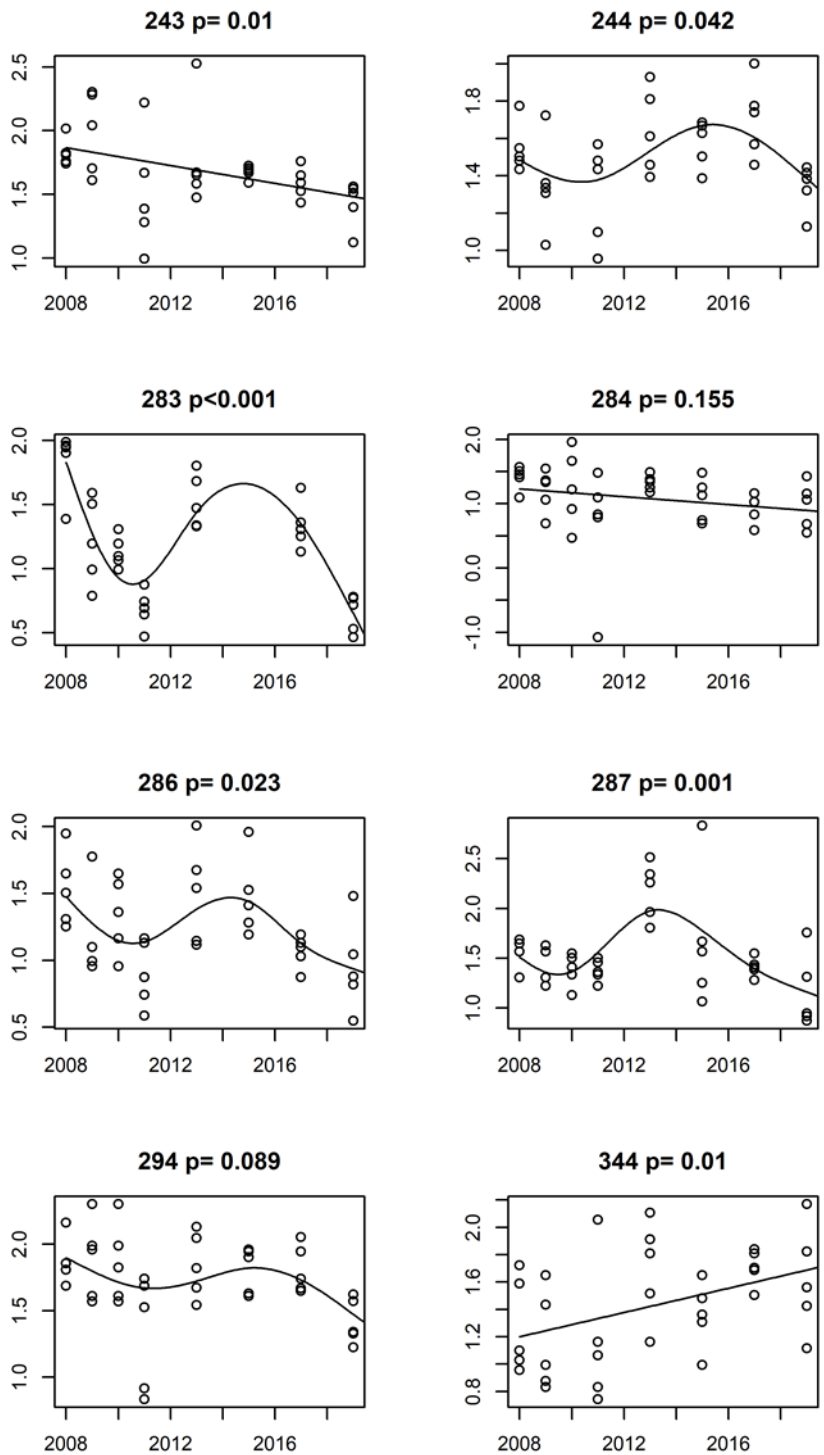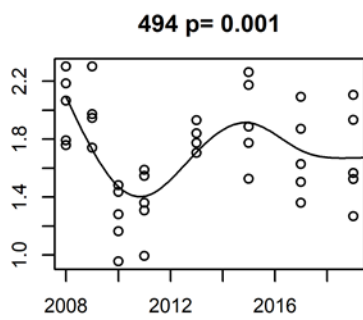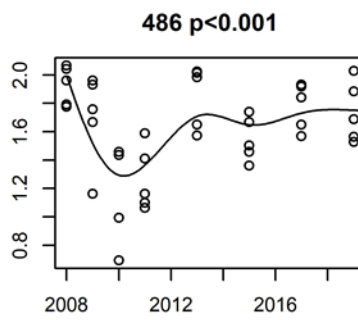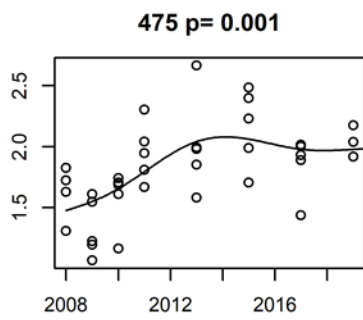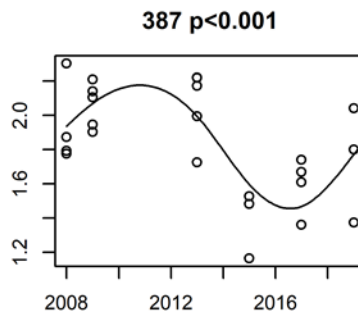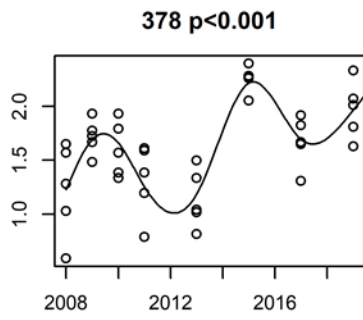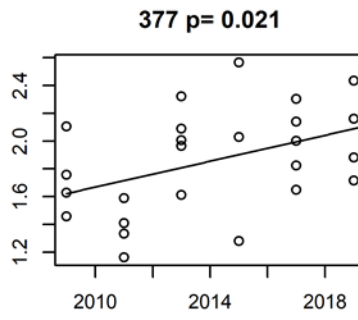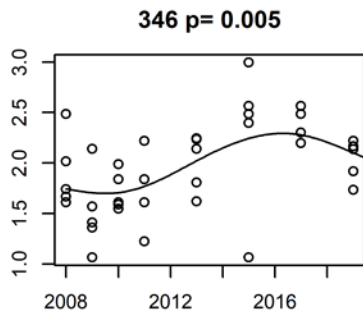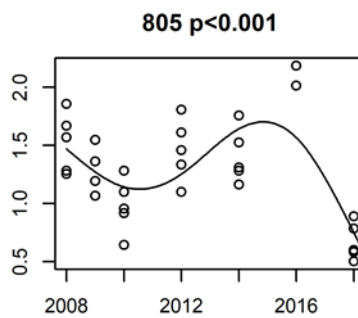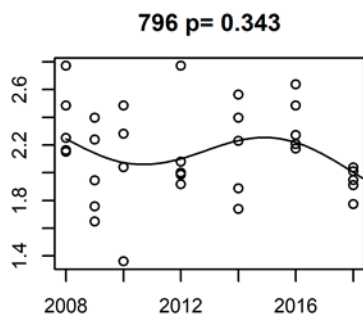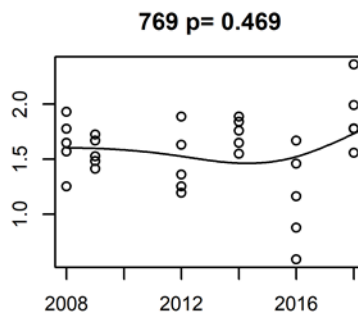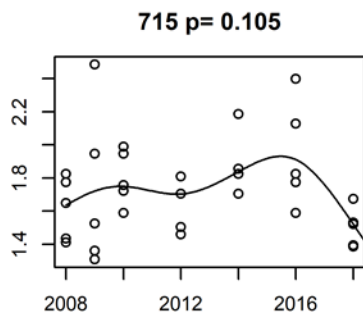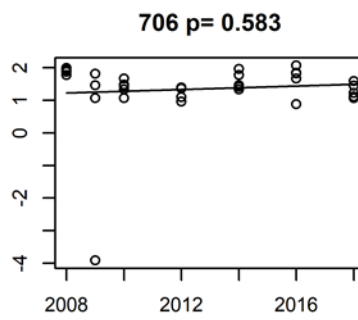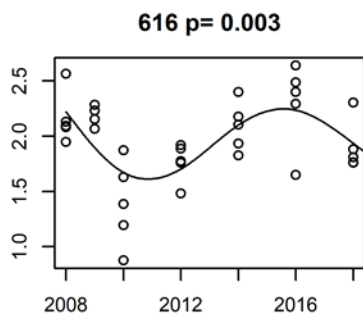
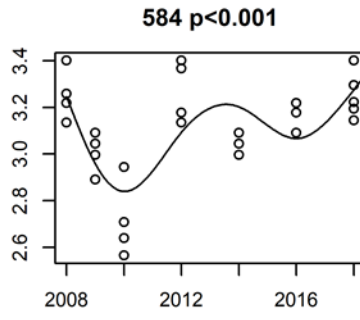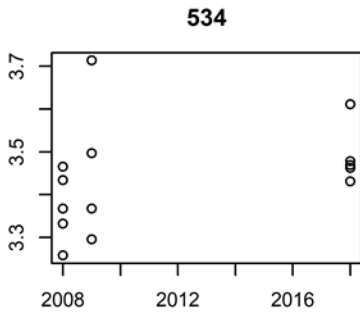**Figure B.4.22.3 Scatterplots of Ln zinc residues in the liver of dab at individual stations around England. The solid black line shows trends from a GAM as a function of time (diagrams courtesy of Cefas)**
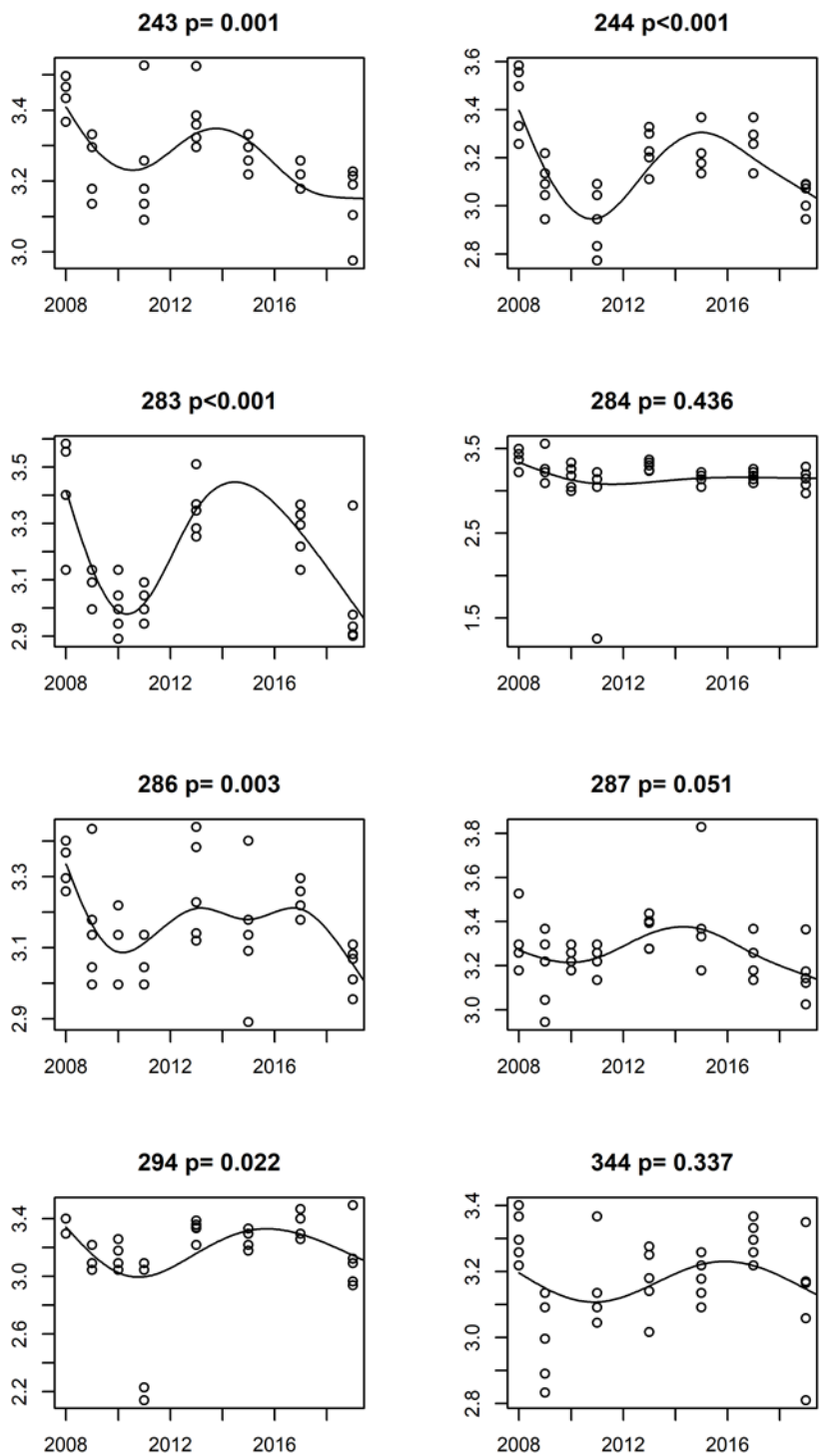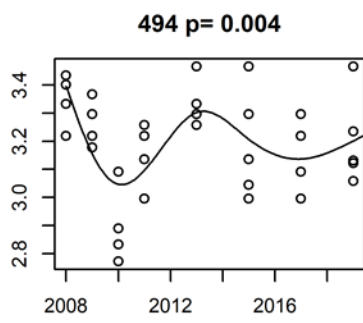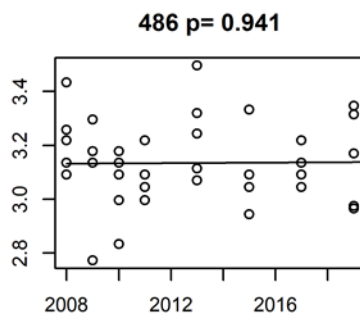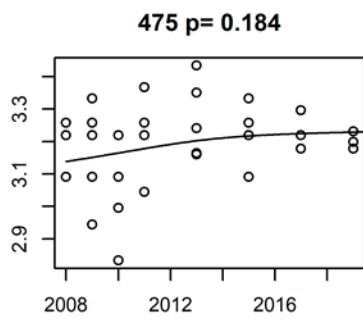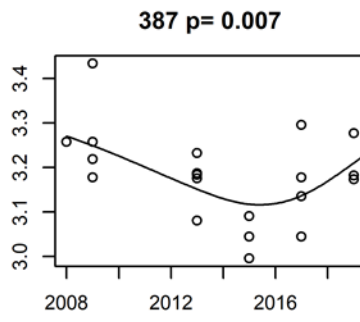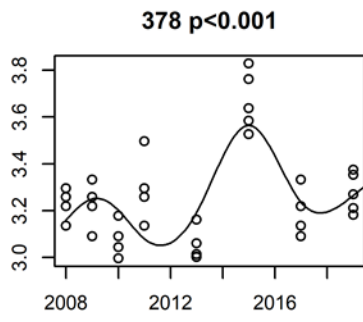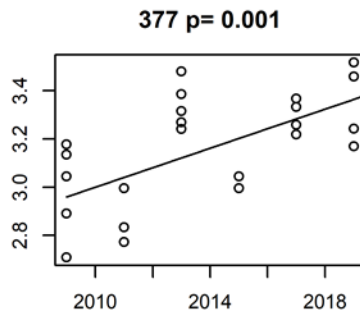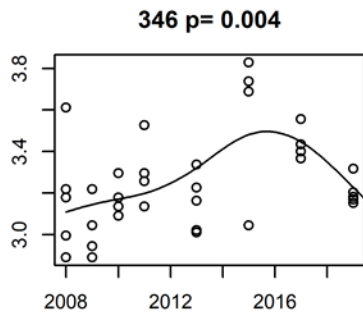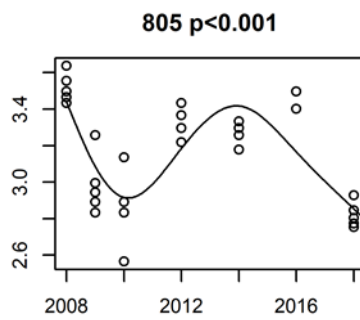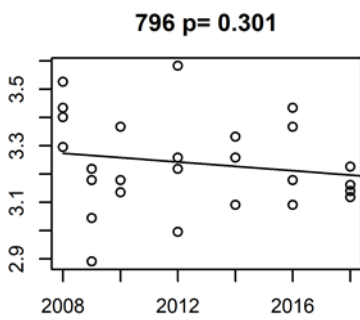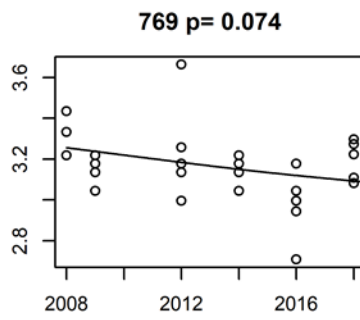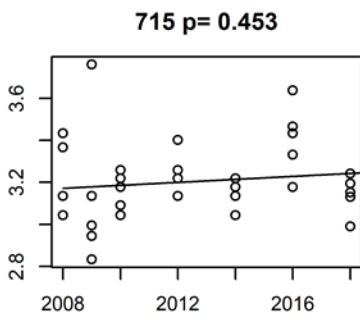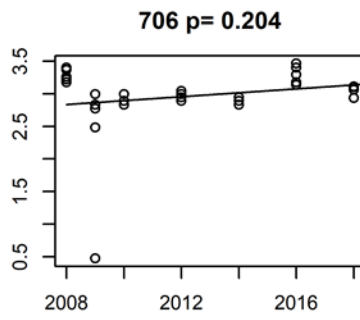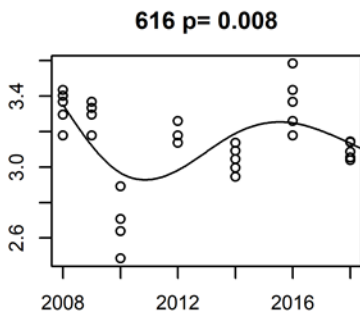
Appendix C Water and Abandoned Metal Mines Programme sites map

Figure C.1 Map showing the Water and Abandoned Metal Mines programme sites

▲ WAMM sampling sites
—— Scottish and Welsh border

List of abbreviations and acronyms

25-YEP   25-Year Environment Plan

ABC   ambient background concentration

APHA   Animal Plant Health Agency

ATSDR   Agency for Toxic Substance and Disease Registry

BODC   British Oceanographic Data Centre

$CaCO_3$ calcium carbonate

Cd   cadmium

Cefas   Centre for Environment, Fisheries and Aquaculture Science

CEMP   Coordinated Environmental Monitoring Programme (OSPAR)

CSIP   Cetaceans Strandings Investigation Programme

Cu   copper

CUOP   Cardiff University Otter Project

Defra   Department for Environment, Food and Rural Affairs

DOC   dissolved organic carbon

EAC   environmental acceptable concentration

EC   European Commission

EQS   environmental quality standard

GAM   generalised additive model

Hg   mercury

ICES   International Council for the Exploration of the Sea

IoZ   Institute of Zoology

JNCC   Joint Nature Conservation Committee

KW   Kruskal–Wallis statistic

Ln   natural logarithm

LoD   limit of detection

M-BAT	Metals Bioavailability Assessment Tool
MSFD	Marine Strategy Framework Directive
Max	maximum
MIME	OSPAR Working Group on Monitoring and on Trends and Effects of Substances in the Marine Environment
Min	minimum
Ni	nickel
OSPAR	The Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR Convention)
p	p-value/probability
Pb	lead
PBDE	polybrominated diphenyl ether
PBMS	Predatory Bird Monitoring Scheme
PBT	persistent, bioaccumulative and toxic
PCB	polychlorinated biphenyl
PFOS	perfluorooctanesulfonic acid
Q1	lower interquartile range
Q3	upper interquartile range
QS	quality standard
QSsec pois	secondary
r_s	Spearman's rank correlation coefficient
SD	standard deviation
SGAR	second-generation anticoagulant rodenticide
TEQ	toxic equivalent
UKCEH	UK Centre for Ecology and Hydrology
UK-SCAPE	UK Status, Change and Projections of the Environment
UKTAG	UK Technical Advisory Group

WAMM   Water and abandoned metal mines

WIIS   Wildlife Incident Investigation Scheme

Zn   zinc

Glossary

Congener structure   One of a group of substances that are related in terms of origin and structure

Dioxin-like PCBs   PCBs which have the same toxic action as 2,3,7,8-tetrachlorodibenzo-p-dioxin – the most toxic chlorinated dioxin congener – albeit at different potencies.

False positive   An error where a result indicates the presence of a condition when it is not there.

False negative   An error where a result indicates the absence of a condition when it is there.

ICES-7   Seven PCBs designated by the ICES as congeners commonly found the environment and therefore an indicator of the degree of contamination. The group include PCB118 which is the most toxic of the seven; without this congener the group is commonly referred to as the ICES-6. PCB118 is also one of a group of 12 PCBs known as dioxin-like PCBs.

p-value   Estimated probability of rejecting the null hypothesis. In the case of the indicator, the p-value is used to help assess trends over time and the null hypothesis is that there is no change in concentration over time. The significance level of 5% is selected – this is an arbitrary value – and if p is less than 0.05 it given reasonable support to the alternative hypothesis that there is a trend.

Stratified sampling   Where a subset of data proportionally reflects any different groups in the full dataset.

TEQ   A system of toxic equivalents used to derive a quantity of polychlorinated dibenzo-*p*-dioxins, polychlorinated dibenzofurans and PCB congeners as a single value based on the relative toxicity of all the congeners to the most harmful derivative, 2,3,7,8-tetrachlorodibenzo-p-dioxin. There are two systems commonly used: I-TEQ adopted by NATO in 1989 and WHO-TEQ which was published in 1998 and updated in 2005. We have used the TEQs for the PCBs only here to express concentrations in the environment in a way that is proportional to their toxicity.

Trophic level   A position within a hierarchy of levels that represents the feeding relationships of all organisms within an ecosystem; a stage in a food chain.

Wet weight   Refers to the sample it is received, regardless of whether it is a whole organism or parts of the organism.

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