

Working with Natural Processes – Evidence Directory

Appendix 2: Literature review

SC150005

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

Acting to reduce the impacts of a changing climate on people and wildlife is at the heart of everything we do.

We reduce the risks to people, properties and businesses from flooding and coastal erosion.

We protect and improve the quality of water, making sure there is enough for people, businesses, agriculture and the environment. Our work helps to ensure people can enjoy the water environment through angling and navigation.

We look after land quality, promote sustainable land management and help protect and enhance wildlife habitats. And we work closely with businesses to help them comply with environmental regulations.

We can't do this alone. We work with government, local councils, businesses, civil society groups and communities to make our environment a better place for people and wildlife.

Published by:

Environment Agency, Horizon House, Deanery Road, Bristol, BS1 5AH

http://www.gov.uk/government/organisations/environment-agency

© Environment Agency - October, 2017

All rights reserved. This document may be reproduced with prior permission of the Environment Agency.

Email: fcerm.evidence@environment-agency.gov.uk

Further copies of this report are available from our publications catalogue: http://www.gov.uk/government/publications

or our National Customer Contact Centre: T: 03708 506506

Email: enquiries@environment-agency.gov.uk

Author(s):

Rachelle Ngai, Mark Wilkinson, Tom Nisbet, Robert Harvey, Stephen Addy, Lydia Burgess-Gamble, Steve Rose, Steve Maslen, Alex Nicholson, Trevor Page, Jennine Jonczyk and Paul Quinn

Dissemination status:

Publicly available

Keywords:

Flood and Coastal Erosion Risk Management, Working with Natural Processes, Natural Flood Management

Research contractor:

Lead contractor: JBA Consulting Consortium delivery team: Forest Research, James Hutton Institute, Lancaster University and Newcastle University

Environment Agency's Project Manager: Lydia Burgess-Gamble

Theme Manager:

Jacqui Cotton - Policy Strategy and Investment

Collaborator(s):

Defra, English Severn and Wye RFCC, Forestry Commission, HR Wallingford, Natural England, Natural Resources Wales, River Restoration Centre, SEPA, Woodland Trust

Project number: SC150005

Evidence at the Environment Agency

Scientific research and analysis underpins everything the Environment Agency does. It helps us to understand and manage the environment effectively. Our own experts work with leading scientific organisations, universities and other parts of the Defra group to bring the best knowledge to bear on the environmental problems that we face now and in the future. Our scientific work is published as summaries and reports, freely available to all.

This report is the result of research commissioned by the Environment Agency's FCRM Directorate and funded by the Joint Flood and Coastal Erosion Risk Management Research and Development Programme. The programme is a joint collaboration between the Environment Agency, Defra, Natural Resources Wales and the Welsh Government. It conducts, manages and promotes flood and coastal erosion risk management research and development.

You can find out more about our current science programmes at https://www.gov.uk/government/organisations/environment-agency/about/research

If you have any comments or questions about this report or the Environment Agency's other scientific work, please contact research@environment-agency.gov.uk

Professor Doug Wilson

Director, Research, Analysis and Evaluation

Executive summary

This literature review summarises the state of knowledge on Working with Natural Processes (WWNP) to reduce flood and coastal erosion risk. Much research has already been carried out on this topic but this is the first time it has been synthesised in one location.

Chapters 2 to 5 contain 4 separate literature reviews which cover:

- river and floodplain management
- woodland management
- run-off management
- coast and estuary management

The 4 reviews look in detail at the individual interventions listed in the table below.

Chapter 2. River and floodplain management	Chapter 3. Woodland management	Chapter 4. Run-off management	Chapter 5. Coast and estuary management
River floodplain restoration	Wider catchment woodland	Land and soil management practices	Managed realignment
Wetland restoration	Cross-slope woodland	Moorland grip blocking	Saltmarsh, mudflat and dune restoration
Remeander rivers	Floodplain woodland	Gully blocking	Beach recharge/ nourishment
Leaky dams/woody dams	Riparian woodland	Run-off attenuation features	Beach bypassing

The reviews establish:

- What we know about:
 - Effect on flood flows, flood peaks and storage
 - Effect on fine sediment and channel conveyance
 - How long does it take to be effective?
 - How does it perform from an engineering perspective?
 - What's its design life?
 - How effective is it at different catchment scales?
 - How effective is it in different watercourse typologies or different catchment geologies?
 - How do you design and construct it so it is most effective?
 - How do you monitor its effectiveness?
 - How frequently does it need to be maintained?
- What we don't know the research gaps

- Costs and benefits
- Multiple benefits

The outcomes of these reviews are summarised alongside case study examples in the <u>Evidence Directory</u>. The purpose of this appendix is to enable those using the Evidence Directory to obtain a greater level of detail on specific areas of interest.

Acknowledgements

We would like to thank SEPA (Heather Forbes) and the Woodland Trust (Christine Reid) for contributing financially to this project.

We are also very grateful to our Project Board and Project Steering Group – from Defra, English Severn and Wye Regional Flood and Coastal Committee, Forest Research, HR Wallingford, Natural England, Natural Resources Wales, the Rivers Restoration Centre, SEPA and the Woodland Trust – who have reviewed and commented on drafts of this literature review.

A big thank you to our external peer reviewers – Angela Gurnell (Queen Mary College, London), Gareth Old and Mike Acreman (CEH), Joseph Holden (Leeds University) and Nigel Pontee (CH2M Hill) – who have all provided invaluable comments and suggested improvements to this document.

This <u>evidence base</u> is dedicated to the memory of our friend and colleague Duncan Huggett, whose pioneering work and dedication to the field of Natural Flood Management has had a significant impact on the development of the policy, science and practice which underpins this report.



Duncan Huggett addressing the Flood and Coast Conference 2017 (Source: Flood and Coast Conference 2017)



Chapte	er 1. Introduction	1
1	Introduction	2
1.1	Project and report objectives	2
1.2	Scope of the review	2
1.3	Methodology	3
Chapte	er 2. River and floodplain management	4
2	River and floodplain management	5
2.1	Introduction	5
2.2	River restoration	6
2.3	Floodplain and floodplain wetland restoration	17
2.4	Leaky barriers	29
2.5	Offline storage areas	47
Chapte	er 3. Woodland management	59
3	Woodland management	60
3.1	Introduction	60
3.2	Catchment woodland	61
3.3	Cross-slope woodland	80
3.4	Riparian woodland	84
3.5	Floodplain woodland	90
Chapte	er 4. Run-off management	96
4	Run-off management	97
4.1	Introduction	97
4.2	Soil and land management	106
4.3	Land and headwater drainage management	139
4.4	Run-off pathway management	165
Chapte	er 5. Coast and estuary management	193
5	Coast and estuary management	194
5.1	Introduction	194
5.2	Saltmarshes and mudflats	196
5.3	Sand dunes	213
5.4	Managed realignment and regulated tidal exchange	224
5.5	Beach recharge/nourishment and sand scaping	249
5.6	Sediment bypassing	262
Refere	nces	270
List of	abbreviations	320

vii

List of tables and figures

Table 1.1 Table 2.1 Table 4.1 Table 4.2 Table 4.3 Table 4.4 Table 4.4 Table 4.5 Table 4.6 Table 5.1 Table 5.2 Table 5.3 Table 5.4 Table 5.5 Table 5.6 Table 5.7	WWNP measures covered in this literature review Summary of example UK washland projects Soil responses after the introduction of no tillage (within 5 or fewer seasons) Relative agronomic advantages and disadvantages of no tillage in Europe Reduction in pollution in the three-tiered pond monitored as part of the Cheviot Futures project Terminology used for run-off pathway management measures Description of farm ponds Estimate of physical storage achieved and costs at Nafferton Farm Categorisation of broad types of coastal FCERM interventions with respect to WWNP Sediment recharge schemes in Great Britain Ecosystem services valuation of Alkborough Flats managed realignment scheme Ecosystem services valuation of Steart Managed Realignment Scheme Managed realignment schemes in Great Britain Managed realignment schemes in western Europe Regulated tidal exchange schemes in Great Britain	2 54 114 126 158 165 167 178 197 209 231 233 239 242 245
	Typical beach slopes for various mean sediment sizes	207
Figure 2.1 Figure 2.2	Summary of process-based aims and associated techniques of river restoration Logie Burn in Aberdeenshire before (left) and after (right) reconnection in 2011 of the old channel wh was disconnected in the 1960s due to channelisation	7 lich 8
Figure 2.3 Figure 2.4	Estimated value of ecosystem service benefits accrued over a 25-year period on the Rottal Burn in	9
Figure 2.5 Figure 2.6	Angus, Scotland Illustration of a floodplain from the NWRM.eu website Floodplain reconnection through embankment lowering on the upper River Dee near Braemar, Aberdeenshire, October 2015 (© Anke Addy)	12 18 23
Figure 2.7	Impact of floodplain restoration at Easter 1998 at Clifton Hampden Bridge (River Thames, Oxfordshire)	26
Figure 2.8	A natural in-stream wood accumulation caused by the collapse of a tree in the Logie Burn catchment Aberdeenshire (© Steve Addy)	:, 30
Figure 2.9 Figure 2.10	A flow restrictor measure during normal flows (left) and a flood (right) (© Mark Wilkinson) A series of flow restrictors located in a small upland gully in the Bowmont catchment in the Scottish Borders(© Steve Addy)	32 35
Figure 2.11	Constructing a flow restrictor in the Eddleston catchment (© Tweed Forum)	37
Figure 2.12 Figure 2.13	Engineered log jam with planted trees in the middle of the structure on a gravel bar in the Bowmont catchment, Scottish Borders (© Steve Addy)	39 42
Figure 2.14	Wood placement measure (two crossed trunks) in the Belford Burn, Northumberland (© Mark Wilkinson)	44
Figure 2.15	Decision support flow chart to help decide whether to install a leaky dam	45
Figure 2.16	Hydraulic matrix classifying washlands by degree of hydraulic control	48
Figure 2.18	Impact of different configurations of ponds along a hypothetical stream reach during July 2009 flood (left) and March 2010 flood (right)	49 49
Figure 2.19 Figure 2.20	Beam Parkland natural capital account Offline storage pond located on a floodplain buffer zone in the Belford catchment, Northumberland, during a flood event (top papel) and after a flood event (bottom papel)	52 57
Figure 4.1	Storm hydrograph elements	98
Figure 4.2	Flow pathways	99
Figure 4.3	Three zones of hydrological activity controlling flow pathways	99
Figure 4.4	Hillslope hydrology response types to incident rainfall events: (A) infiltration excess; (B) subsurface stormflow and saturation excess flow where the water table has risen in response to rainfall (dotted b line); and (C) perched water table causing shallow interflow and water logging or drain flow removing surplus flow	olue J 100
Figure 4.5	An intensively farmed arable soil with underdrainage, secondary drainage (mole drains), a compacte soil layer and surface run-off flow pathways	d 101
Figure 4.6	Example of soil drainage concepts used by farmers as presented in guidance from the Agricultural & Horticultural Development Board	102
Figure 4.7	Impact of Drainage on Peat Hydrology	104
Figure 4.8	Synchronisation and combination of subcatchment hydrographs	105
Figure 4.9	Equipment for subsoiling and topsoil loosening	109
rigure 4.10	rundamental soil physical properties and surface crusting can affect the relative proportions of overla flow, lateral throughflow and vertical infiltration	and 110
Figure 4.11	Plot trial layout at Loddington	120
Figure 4.12	Conceptualisation of the Hood system that links land management changes to changes in flood risk	124
Figure 4.13 Figure 4.14	Headlands and wheelings causing run-off (left) and attempts to use cross-slope cultivation to disrupt	13/
Figure 4.15	Operation of hump cross and channel cross drains	140
Figure 4.16	A known flow pathway on a track is drained and piped to a nearby field area (left) and a cross drain directing surface flow onto a grass field (right)	142

Figure 4.17	Conversion of a conventional ditch to a two-stage ditch	143
Figure 4.18	Ditch vegetation attenuating flow (left) and a vegetated ditch (right)	144
Figure 4.19	Willow growth within a ditch (left) and living barriers (right will establish a permanent self-maintained	111
Figure 4 20	leaky structure	144
Figure 4.20	(right)	145
Figure 4.21	Widening a ditch and adding a recycled leaky plastic barrier at Nafferton Farm (left) and a widened c	litch
i iguio 1.21	intercepting a known overland flow pathway (part of the MOPS project) (right)	145
Figure 4 22	Three-tiered sediment trap in Netherton Burn Northumberland (left) and the impact of the ponds in	1.10
rigaro 1.22	attenuating the flow (right)	146
Figure 4 23	Lowland ditch management options: vegetation cutting (left) and sediment removal (right)	150
Figure 4 24	Bromoton study where leaky barriers are being trialled and simulated	151
Figure 4 25	The Great Eep showing (left) a typical sluice for maintaining the water level in a ditch and (right) an	
rigaro 1.20	example of steep sided ditch that has been regraded	151
Figure 4.26	Types, inputs and outputs of catchment modelling approaches	153
Figure 4 27	1D bydraulic model of a widened and roughened ditch Nafferton Farm study	154
Figure 4.28	Schematic representation of run-off generation in a (recently) blocked grip prior to, early on in or after	r
. iguio iiizo	a rainfall event (left) and during a rainfall event (right)	157
Figure 4.29	A trial leaky dam in the ditch at Nafferton Farm (left) and Belford Burn using a series of wooden flow	
	restrictors (right)	163
Figure 4.30	Corner of field ponds at Nafferton Farm for local run-off from the field and the farm tracks	169
Figure 4.31	Diagram of a simple sediment trap	170
Figure 4.32	Typical swale design. Left: a well-maintained feature and (right) a less maintained, roughened surface	ce
0	and containing a stone check dam	171
Figure 4.33	Output from RAF-11 in Belford during June 2012 storm event	173
Figure 4.34	Data from the pilot pond during September 2008 storm event	180
Figure 4.35	Output from Farm PLOT identifying the location of RAF-11 in the Belford catchment	184
Figure 4.36	Field treatment train	185
Figure 4.37	An example of the detailed design for ponds	186
Figure 4.38	Water balance for permanent ponds	186
Figure 4.39	Field bund storing water in a storm	187
Figure 4.40	Pilot pond at Belford full of water following a storm event in September 2008	188
Figure 4.41	Leaky nature of the pilot pond at Belford	188
Figure 4.42	Example design outline for a sediment trap	189
Figure 4.43	Typical design for a swale for a farm	190
Figure 5.1	Wave reduction by 4 different habitat types	207

Chapter 1. Introduction



Beam Washlands, Dagenham (source: Environment Agency)

1 Introduction

1.1 Project and report objectives

Working with Natural Processes (WWNP) is a form of flood risk management that can be implemented on hill slopes, rivers, floodplains, estuaries and coasts.

This literature review was written to underpin the preparation of the WWNP <u>Evidence</u> <u>Directory</u>, which has been developed to help flood and coastal erosion risk management (FCERM) authorities develop FCERM projects that include WWNP to reduce flood risk.

Considerable research has already been conducted on this topic but this is the first time it has been synthesised in one location.

This chapter outlines the purpose of this literature review and the approach taken to synthesise a vast array of academic and grey literature. The following 4 chapters contain separate literature reviews covering:

- Chapter 2 River and floodplain management
- Chapter 3 Woodland management
- Chapter 4 Run-off management
- Chapter 5 Coast and estuary management

The content of this literature review is summarised within the <u>Evidence Directory</u>, which is supported by <u>one-page summaries</u>, case study vignettes and detailed case studies.

1.2 Scope of the review

1.2.1 What topics does this report cover?

This report contains 4 separate literature reviews in chapters 2 to 5. Within each chapter, a range of different WWNP measures is covered in individual sub-sections (see Table 1.1).

Chapter 2. River and floodplain management	Chapter 3. Woodland management	Chapter 4. Run-off management	Chapter 5. Coast and estuary management
River floodplain restoration	Wider catchment woodland	Land and soil management practices	Managed realignment
Wetland restoration	Cross-slope woodland	Moorland grip blocking	Saltmarsh, mudflat and dune restoration
Remeander rivers	Floodplain woodland	Gully blocking	Beach recharge/ nourishment
Leaky dams/woody dams	Riparian woodland	Run-off attenuation features	Beach bypassing

Table 1.1	WWNP measures covered in this literature review
-----------	---

For each of the measures in each chapter the literature review covers the following topics:

- What we know about:
 - Effect on flood flows, flood peaks and storage
 - Effect on fine sediment and channel conveyance
 - How long does it take to be effective?
 - How does it perform from an engineering perspective?
 - What's its design life?
 - How effective is it at different catchment scales?
 - How effective is it in different watercourse typologies/catchment geologies?
 - How do you design and construct it so it is most effective?
 - How do you monitor its effectiveness?
 - How frequently does it need to be maintained?
- What we don't know the research gaps
- Costs, benefits and multiple benefits

Each chapter has been subject to an external peer review by a lead expert within the specific field of WWNP.

1.3 Methodology

This report was developed through a review of existing academic and grey WWNP literature both internationally and from the UK. Relevant academic papers, policy reports and grey literature were initially identified using the existing knowledge of the literature across JBA Consulting, its consortium, the Environment Agency and wider members of the Project Steering Group.

The following topic-leads were established to work on the development of the literature reviews for specific chapters:

- Chapter 2 River and floodplain management
 - Topic lead Mark Wilkinson
 - Additional support Stephen Addy
- Chapter 3 Woodland management
 - Topic lead Tom Nisbet
- Chapter 4 Run-off management
 - Topic lead JBA Consulting (Rachelle Ngai, Steve Maslen and Steve Rose)
 - Additional support Paul Quinn, Alex Nicholson and Jennine Jonczyk
- Chapter 5 Coast and estuary management
 - Topic lead Robert Harvey

Chapter 2. River and floodplain management



2.2 River restoration



2.3 Floodplain and floodplain wetland restoration



2.4 Leaky dams



2.5 Offline storage areas

2 River and floodplain management

This chapter forms the literature review for river and floodplain WWNP interventions.

Sections 2.1 to 2.5 cover in detail each of the different WWNP interventions reviewed:

- 2.2 River restoration
- 2.3 Floodplain and floodplain wetland restoration
- 2.4 Leaky barriers
- 2.5 Offline storage areas

2.1 Introduction

2.1.1 What is WWNP in the context of river and floodplain management?

This section reviews the evidence concerning the role of river and floodplain management in managing flood risk. It considers 4 different types of intervention:

- river restoration
- river floodplain restoration
- leaky dams
- offline storage areas

Restoring the natural processes and features within rivers can provide a wide range of benefits for the environment and people. With regard to FCERM benefits, restoration can increase hydraulic roughness and morphological complexity. Together this can reduce water velocities and help to reconnect floodplains and increase temporary water storage, which in turn attenuates floods. The FCERM benefits of river restoration can be maximised when combined with the restoration of floodplains, adjacent wetlands and off-channel areas.

Floodplains and floodplain wetlands can be restored or created to store large volumes of water for flood risk and ecological benefits. Floodplain restoration aims to restore the hydrological connection between rivers and floodplains, so that floodwaters inundate the floodplains and store water during times of high flows. This can involve removing flood embankments and other barriers to floodplain connectivity. Wetlands are dynamic and changing habitats that include fens, dune slacks, grazing marsh and swamp, upland and lowland peat bog, reedbed and saltmarsh, wet woodland, wet grassland and wet heathland. This chapter deals only with floodplain wetlands. The other types of wetland are covered in the other chapters.

Leaky barriers usually consist of pieces of wood – occasionally combined with some living vegetation – that accumulate in river channels as well as on river banks and floodplains.

Although the word 'barrier' evokes thoughts of hard engineering, leaky barriers occur naturally along rivers as a result of trees falling locally into watercourses through snagging of natural wood or occasionally due to beaver activity. Similar structures can also be engineered by humans to restore rivers and floodplains, and to help slow and store water.

Offline storage areas are floodplain areas that have been adapted to retain and attenuate floodwater in a managed way. They usually require the construction of a containment bund, which increases the amount of water that can be stored on a floodplain and may also require an inlet, an outlet and potentially a spillway mechanism. This chapter covers small to medium scale offline storage areas. It does not included engineered flood storage areas, which are typically online and built to reservoir safety standards with an outflow controlled by flow control devices.

2.2 **River restoration**

Human actions have had a widespread effect on rivers over the centuries. Examples of pressure that have affected rivers include (Holmes and Raven 2014):

- land use change or dam installation which affect the processes within a reach
- modifications by straightening, enlarging, embanking and reinforcing the channel

As a result, river restoration has become a prominent strand of river management to reverse these impacts which date back in the UK to the 1980s (Smith et al. 2014).

River restoration can be described as:

'the return of the physical (for example, sediment, morphology and flow regime) character and habitats of a river channel (including wetted areas, riverbanks and floodplains) to a more natural state' (adapted from Roni and Beechie 2012).

This is achieved by applying indirect or passive (for example, removal of dams or dealing with human disturbances by changing riparian land management practices) and active or direct techniques (using physical actions to restore physical processes and features). Importantly, this does not mean returning a river to its pre-disturbance state as this can be practically impossible, costly and is usually undesirable (Wharton and Gilvear 2007, Dufour and Piégay 2009).

Increasingly, restoring processes that sustain a natural, self-sustaining river morphology and habitat mosaic are being promoted (Beechie et al. 2010) (Figure 2.1). This emphasises restoring process connectivity along the river network from upstream to downstream, laterally between the river and its floodplain, and vertically with any underlying alluvial aquifer (Kondolf et al. 2006), rather than imposing some predetermined channel form that may not reflect current, past or likely future processes (Kondolf 2006).

River restoration actions may be indirect or direct (Figure 2.1). Indirect methods are applied beyond the reach that is to be restored and are aimed at reinstating more natural processes. Examples include:

- changes in land use and management to moderate the flow of water, sediment, organic material and organisms from the catchment to the river network
- modification of flow and sediment releases from dams
- the removal of dams and other barriers to reinstate more natural flows through the river network to the restoration site

Direct methods are applied to increase the potential of a reach to respond naturally to processes. Examples include (Piégay et al. 2005):

- the removal of bank and bed reinforcement
- the removal or setting back of embankments to allow the form and position of the river channel to adjust to received processes within an erodible river corridor

Reinstatement of river processes, erodible channel boundaries and a cessation or reduction in sediment and vegetation management allow physical processes and vegetation to interact and rapidly achieve adjustments in channel morphology (Corenblit et al. 2007, Gurnell, 2014, Gurnell et al. 2016). It is important to recognise that such interactions and adjustments can occur quite quickly, even in very low energy rivers (Gurnell and Grabowski 2016). Furthermore, in cases where river energy levels are sufficient, simply halting channel and hard engineering maintenance and allowing the river to undermine artificial lateral constraints may be sufficient to initiate river self-recovery (Parsons and Gilvear 2002).



Figure 2.1 Summary of process-based aims and associated techniques of river restoration

Source: from Addy et al. (2016)

Process-based river restoration approaches have several advantages over stream rehabilitation approaches that attempt to mimic particular river channel forms or habitat assemblages through engineering (Beechie et al. 2010, Mainstone and Holmes 2010).

- They tackle the root causes of the problem rather than the symptoms.
- By relying on natural processes to do the work, physical features and habitat more in keeping with a given river are likely.
- They are often more cost-effective and sustainable in the long term.

Where societal constraints exist that limit the restoration of natural processes, direct techniques are often used to manipulate or create river channel forms to achieve a desired state. Such solutions can be aesthetically pleasing and can focus on ensuring appropriate channel conveyance and stability or the presence of particular habitats, but they constitute rehabilitation rather than true restoration measures, since they do not depend on a free interaction between processes and forms (Shields et al. 2003). Furthermore, if the rehabilitated forms are to persist, they are likely to require significant maintenance. Reconnection of old river channels or the creation of new ones that mimic historical channel patterns are a widely used example of measures that are often employed where channel and habitat forming processes have been compromised or the timescales for self-recovery are unacceptably long (Figure 2.2).

Traditionally, river restoration has been carried out for fishery enhancement, habitat restoration and biodiversity conservation motives (Gilvear et al. 2012, Griffin et al. 2015, Wohl et al. 2015). Increasingly, however, the roles of river restoration and rehabilitation as a means of reducing flood risk in addition to meeting habitat and biodiversity or other objectives is being recognised (Wharton and Gilvear 2007, Nardini and Pavan 2012, SEPA 2015). In contrast with traditional river management approaches that rely on hard engineering to reduce flooding by increasing the channel conveyance of floods, river restoration projects often aim to reduce flood risk through increasing floodwater storage and attenuation effects (Gilvear et al. 2012).



Figure 2.2 Logie Burn in Aberdeenshire before (left) and after (right) reconnection in 2011 of the old channel which was disconnected in the 1960s due to channelisation

2.2.1 Understanding the science

It has long been recognised that the size and form (width, depth, sinuosity, slope) and the bed and bank sediment characteristics of a river channel are dependent upon the prevailing flow regime and the calibre and quantity of sediment supplied to the river and are constrained by the valley gradient (Schumm 1977) (Figure 2.3). Changes in flows of water and sediment lead to adjustments in river channel form. The nature and rate of these

adjustments are highly uncertain because of the many degrees of freedom (for example, channel width, depth, sinuosity, planform and slope) that are available for river channel adjustment and the many feedbacks that occur during these adjustments.

Nevertheless the capacity, width, depth and pattern of naturally adjusting river channels generally reflect properties of the flow and sediment transport regimes (Hey and Thorne 1986, Wharton 1995, Church 2006). For example, river channels typically adjust their dimensions so that, when the water level is at bankfull, the channel can accommodate a flow with a return period (based on analysis of an annual maximum flood series) of 1–3 years, with a modal value of around 1.5 years (Petts and Foster 1985). Continual changes in flow and sediment processes in response to changes (for example, in climate, land cover, and other human pressures and interventions) lead to continuous adjustments in the size and form of river channels (Winterbottom 2000). As a result, most river channels do not reach an equilibrium form over time from the mutual adjustment of these variables. Instead, rivers not only display continuous short-term adjustments (days to years), but also longer term trajectories of adjustment (decades to centuries) during which their size and morphology can change completely, for example, from a braided to a meandering channel pattern.



Figure 2.3 Overview of the key controls on the river channel form

Source: Thorne et al. (1997)

The important influence of vegetation on river channel size, form and adjustments has also been recognised (Solari et al. 2016). In particular, the ability of certain riparian and aquatic plant species to act as physical engineers of river ecosystems leads to the development of characteristic landforms or habitats in and around river channels – including vegetated bars, benches and islands – which facilitate colonisation by other plant species and lead to adjustments in channel form, position and size (Gurnell 2014). Such interactions between river processes, plants and large wood are an important component of river self-restoration, particularly in low energy rivers (Gurnell and Grabowski 2016).

Conveyance capacity can be defined as the ability of a river to convey flow and is controlled by channel size, gradient and hydraulic roughness (Brookes and Shields 1996). Channel capacity depends on the cross-sectional area of the main channel and any side channels that may be present in the case of multi-thread (for example, wandering or anabranching) rivers. Vertical or lateral erosion and the deposition of sediment within a river channel change its size. Conveyance capacity is also influenced by the hydraulic roughness of the channel, which is affected by its form (bedforms, cross-section form, planform) and 'skin' (bank and bed sediment size, aquatic and riparian channel vegetation, presence of large wood) (Buffington and Montgomery 1999, Yarnell et al. 2006, O'Hare et al. 2010). The variability of these sources of hydraulic roughness influences conveyance, and in turn flood risk. Channels with the full range of these natural sources of hydraulic roughness would be expected to more effectively reduce water flow velocities and also encourage connectivity with the floodplain (Keesstra et al. 2012, Thompson et al. 2015). Both of these effects could in turn help to attenuate floods.

Human modification of river channels, either by directly altering channel dimensions and removing channel vegetation, or by indirectly changing controlling processes, can result in major alterations in the attributes of a river (Wohl 2005). Human activities such as land use change, channel sediment management, flood defence engineering or land drainage alter channel dimensions and roughness, and affect the channel's conveyance capacity. Changes in conveyance over time as a result of these activities can in turn influence local flood risk (Slater 2015). Modification of in-channel and marginal terrestrial vegetation communities may further affect the conveyance capacity by altering the hydraulic feedbacks and morphological controls that vegetation exerts (Gurnell 2014).

One of the main causes of river degradation is channelisation. The term 'channelisation' refers to a range of engineering interventions 'for the purposes of flood control, drainage improvement, maintenance of navigation, reduction of bank erosion, or (channel) relocation' (Brookes 1988, CIWEM 2014). Methods of channelisation include (Brookes 1988):

- clearance of pioneer trees, 'weeds' and aquatic plants, silt and trash
- bank protection
- embanking
- straightening
- re-sectioning (widening and/or deepening)

The action of straightening is particularly damaging, since it increases the river channel gradient and simplifies the channel planform, often resulting in bed incision and the need to reinforce the channel banks to prevent erosion. The dredging required to maintain the size and simple form (typically trapezoidal) of enlarged channels leads to a loss of the hydraulic roughness created by natural bank profiles, bed forms and vegetation (CIWEM 2014). Together, these channelisation actions increase flow velocity and reduce time to peak flow, potentially leading to the unintended effect of increasing flooding problems downstream (CIWEM 2014). In addition, the increased stream power associated with reduced roughness and steepened channels may not only lead to erosion within the channelised reach, but also upstream bed incision and sometimes downstream sediment deposition, followed by a sequence of adjustments in channel size and morphology that propagate along the channelised and upstream or downstream channels (Simon 1989, Simon and Rinaldi 2006). As a result of widespread channelisation, actively meandering and multi-thread channels are now very rare. Nearly 9,000km of the river channel network in England and Wales are believed to have been artificially channelised (Brookes et al. 1983).

Restoring natural channel morphologies and processes may increase the attenuation of high flows. Natural flow and sediment transport processes create smaller channel sizes than artificially enlarged channels and more influential sources of hydraulic roughness (natural bed forms, bank profiles and vegetation) that reduce flow velocities (Buffington and Montgomery 1999). Natural channels also tend to have greater connectivity to their floodplain in part due to their lower conveyance capacity and bank heights than artificially altered channels. This could increase temporary floodwater storage and further attenuate flood peaks. In addition, free lateral connectivity to floodplains can aid capture of sediments

and thus help to mitigate sediment-related problems further downstream (Thompson et al. 2015). The effect of free floodplain connectivity on attenuation is maximised when both restoration of channels and floodplain areas (see Chapter 3) occurs in combination. For example, a restored and reconnected floodplain with characteristic woodland will enhance temporary floodwater storage and hydraulic roughness more effectively than a degraded one (Thomas and Nisbet 2006).

One of the most important characteristics of naturally functioning rivers is their ability to reshape themselves. Allowing natural rivers freedom to adjust through bank erosion by removing lateral constraints such as revetments is important in that it allows them to maintain their capacity to accommodate sediment and flow regimes under climate change (Raven et al. 2009). Over time meanders can develop in lower gradient settings and their sinuous courses can increase the contact length with floodplains (Sholtes and Doyle 2011), and create hydraulic feedbacks including super-elevation of the water surface at the points of bends that results in floods in more frequent hydrological connection to the floodplain. River widening, which is often accompanied by removing lateral constraints, has often been practised in Europe and can help to initiate natural erosion and deposition processes (Rohde et al. 2006, Leite Ribeiro et al. 2015). Approaches are available for defining the optimal zone in which to allow free river movement (Piégay et al. 2005).

Restoration of natural in-channel and marginal terrestrial vegetation communities can also create important hydraulic and geomorphic feedbacks (Eekhout et al. 2014, Gurnell 2014, Solari et al. 2016) that in turn enhance hydraulic roughness and floodplain connectivity. Bank erosion can cause feedbacks such as the input of trees and other vegetation material, which may further increase channel roughness and attenuation effects (Gregory et al. 1985). The increase in channel sources of hydraulic roughness and bank roughness through the development of natural vegetation communities can also help to dissipate energy during floods and in turn reduce sediment-related problems (Keesstra et al. 2012).

2.2.2 Confidence in the science

Confidence in the relationships between channel morphology, roughness and conveyance as summarised above is good. However, the evidence on the effect of river restoration through manipulating river form or through restoring processes on FCERM has previously been regarded as limited (Wharton and Gilvear 2007), and with regard to empirically based studies, this is still the case. This reflects the limited attention to this particular facet of river restoration, which is perhaps not as well known as the potential habitat and biodiversity benefits. It could also reflect the difficulty of isolating and quantifying the attenuation effect of channel restoration (Sear et al. 2006). However, empirical studies that consider the impact of river restoration on flow hydraulics, sediment dynamics and morphology are becoming increasingly available (Florsheim et al. 2006, Eekhout et al. 2014). Such studies contribute to understanding change in channel conveyance capacity that could have implications for FCERM. Other relevant studies include the assessment of the hydraulic and morphological effects of rehabilitation measures (Shields et al. 1995, Gardeström et al. 2013, Abel et al. 2016), but also actions that have a major impact on processes such as removing dams or weirs (Costigan et al. 2014, Claeson and Coffin 2015). These studies, combined with the understanding of the science presented above and model-based understanding, provide a starting point to predict and give guidance on the effects of river restoration on FCERM and their uncertainties.

2.2.3 Metrics of cost–benefit and multiple benefits

Metrics on the costs and benefits to society specific to individual river channel restoration projects are often lacking. This is because river channel restoration measures are often

combined with other measures or the importance of valuing costs and benefits is overlooked (Ayres et al. 2014). Furthermore, the individual benefits accrued by restoration are often considered as part of a larger group of benefits, making it difficult to extract separate values for particular services (Ayres et al. 2014).

Predicting and quantifying the impacts of specific river restoration measures is difficult (Ayres et al. 2014). While the precise morphological impacts of restoration are uncertain (Resop et al. 2014), the social benefits are often not considered (Brouwer et al. 2015). However, there are some studies. Before and after monitoring of the local public's perception of the River Skerne restoration in Darlington over 10 years showed that, with long-term community involvement, a positive public reaction and increase in recreational value are possible (Åberg and Tapsell 2013).

Some researchers have used valuation techniques to estimate the monetary value of different services expected from river restoration. For example, Vermaat et al. (2016) showed that the total ecosystem service value in 8 restored European rivers was significantly increased with a positive difference of €1,400 per hectare per year. In the UK, Mellor (2014) estimated the financial returns of different ecosystem service benefits as a result of restoring the Rottal Burn in Angus. This analysis showed that a high return on investment into river restoration was possible over a 25-year period (Figure 2.4). Prediction of the climate change adaptation and flood risk mitigation benefits of the Mayesbrook Park river restoration in north-east London suggested that, for every £1 spent on the project, a long-term return to society of at least £7 would be gained (Natural England 2013). A benefits transfer approach is available to assess river restoration benefits when it is not possible to carry out primary valuation research (Brouwer et al. 2015).



Figure 2.4 Estimated value of ecosystem service benefits accrued over a 25-year period on the Rottal Burn in Angus, Scotland

Notes: Ecosystem service gains from Mellor (2014). The meandering course of the Rottal Burn was restored in 2012.

Restoration often creates rapid improvements in within-reach characteristic processes and habitat (Kronvang et al. 1998, Sear et al. 2006, Nilsson et al. 2015), though ecological responses vary. In an extensive review of river restoration and rehabilitation projects, Palmer et al. (2010) found that the ecological effects of projects were mostly limited. In contrast, the review by Kail et al. (2015) showed that the effect of restoration on biota was positive overall but that the variability of responses was high.

Specific studies have reported a number of positive ecological outcomes. For example, Rohde et al. (2005) showed river widening in Switzerland could increase habitat heterogeneity and enhance establishment of riparian and pioneer plants. Lüderitz et al. (2011) investigated hydromorphological and ecological (macroinvertebrates, fish and macrophytes) responses in restored sections (reconnected backwaters, restoration of multiple channels and establishments of wide buffer zones) of the Upper Main in Germany. They found that restoration resulted in good ecological status as defined under the Water Framework Directive, in contrast to degraded sections of the same river and the Roddach River. In the UK, positive but short term (<3 years) ecological responses have been found at recent river restoration sites. For example, fish abundance responses to river restoration have been shown to be positive on the restored Ryburgh Loop of the River Wensum, Norfolk, and a greater diversity of macroinvertebrates has been found in the restored section of the Rottal Burn, Angus (Addy et al. 2016).

The varied ecological responses in part reflect overriding factors such as the continued pressures from degraded water quality that are not dealt with by the restoration measures or a lack of source populations of monitored species (Palmer et al. 2010, Feld et al. 2011). Ecological responses also inevitably lag behind morphological responses as the biota colonise what was a previously hostile environment, suggesting that more sophisticated monitoring schemes are needed to capture ecological responses (Nilsson et al. 2015). The work of Jähnig et al. (2010) showed that active restoration measures over relatively short scales (several hundred metres) in 26 European countries improved river and floodplain habitat diversity but that invertebrate responses were limited. This highlighted the importance of the scale and type of restoration being appropriate to ensure a successful ecological response.

Knowledge of the ecological effects of physically restoring riverbanks is limited, but the response to restoring the characteristic vegetation of riverbanks has been assessed. In areas used for agriculture, improving riparian vegetation communities can benefit in-channel macroinvertebrate communities (Feld 2011) and terrestrial beetle species (Stockan et al. 2014).

2.2.4 Effectiveness/performance

Model based

Several model-based studies have been applied at the reach scale to predict the hydraulic and attenuation effects of remeandering or smaller sub-reach scale measures. Using the UNET and HEC-RAS models, Sholtes and Doyle (2011) showed a modelled 1km length of lowland channel restored reinstating a natural meandering planform reduced floods of a small to intermediate magnitude (between a 2 and 50 year return interval) by <1%. They found that attenuation was sensitive in decreasing order of importance to valley slope, channel and floodplain roughness, and channel and valley length. Despite the attenuation response noted, they argued that the small scale of river restoration currently carried out was unlikely to provide quantifiable flood attenuation. This highlights the need for more

widespread restoration of channels and other natural flood management (NFM) aspects covered elsewhere in this review to achieve tangible FCERM benefit.

The hydraulic effects of small-scale manipulation of channel cross-section and bedforms have also been investigated. Using HEC-RAS, Sear and Newson (2004) assessed the hydraulic impacts of a series of restored riffle bedforms on flow stage in a lowland gravel-bed river in Norfolk. The work showed that, at bankfull discharge, water surface elevation was not increased significantly (0.05m on average) over the pre-restoration channel conditions. Downs and Thorne (2000) used a combination of models to predict the effects of channel reprofiling, deflector installation and riparian planting (reeds and trees) on the channel conveyance capacity of a lowland river in Nottinghamshire. It was predicted that these measures would result in about a 10% reduction of conveyance at flows approaching bankfull level. Together, these studies suggest that a minor decrease in conveyance and a slight increase in floodplain connectivity are possible by adding bedforms, increasing riparian vegetation and altering channel cross-sections.

At the reach scale, Keesstra et al. (2012) showed using HEC-RAS that hydraulic roughness was significantly increased under a scenario of natural channel morphology and increased characteristic riparian vegetation cover. Resulting water velocities for a 100-year return interval flood were 41% lower than for a channelised, unnatural channel with no established riparian vegetation, suggesting that restoration of both channels and riverbanks could slow down the passage of flood waves.

At the catchment scale, a number of model-based studies have been applied. Liu et al. (2004) used a distributed hydrological model to predict the effects of increasing meanders and flow resistance on headwater channels in a 400km² catchment. They found an average reduction of peak flows of 14%. Sear et al. (2000) found that reinstating the sinuosity of meandering channels and reducing the channel conveyance capacity of headwater streams reduced the flood height downstream by 20%.

Empirically based

Empirically based studies that focus directly on the attenuation and FCERM benefits of river restoration are lacking. A notable exception is the case study of the hydrological effects of river restoration undertaken in 2004 on lowland streams in the New Forest, Hampshire. Stream reaches were restored through a combination of floodplain reconnection, river morphology restoration and log jam creation to restore habitat and create other benefits.

Remeandering increased river length by 21% and increased log jam frequency by 142% (Kitts 2010). Kitts (2010) found that these measures together resulted in a 21% reduction of flood peak magnitude and a 33% increase in flood peak travel time for flows that were less than 1m³s⁻¹ (equal to a 2-year recurrence interval). Isolating and quantifying the effects of the individual restoration measures was not possible, but all measures are likely to have been influential (Sear et al. 2006). Using pre- and post-restoration monitoring over 3 years at 3 restoration sites in the same catchment, Sear et al. (2006) found that floodplain inundation frequency and duration were increased as a result of remeandering and the reduced channel capacities.

2.2.5 Key case studies

This section presents example case studies of river channel restoration from the UK. In most cases, reducing flood risk was not the main aim of the river restoration measure(s) used and monitoring of the effects on conveyance and attenuation has not been conducted. Furthermore, in some cases several techniques have been used over different scales which

mean that isolating and measuring the effects of a single channel restoration measure is difficult.

Examples of river morphology restoration with FCERM monitoring

- Eddleston Water, Peebles, Scottish Borders. Over 2km of river was remeandered (in addition to tree planting, wetland creation and installation of inchannel wood) in a 50km² catchment. The catchment has a dense hydrological monitoring network and is part of the Interreg Building with Nature project. Owing to the short time span of the monitoring so far (it started in 2011), there are no findings highlighting FCERM benefits.
- New Forest streams (Highland Water and Blackwater), Hampshire. This case study involved the reintroduction of large wood into the channel and reinstating a meandering river planform over nearly 1.5km (Sear et al. 2006). A detailed study of the geomorphic, habitat and hydrological responses was made (Sear et al. 2006, Kitts 2010). This work showed that the combined effects of river restoration and in-channel wood addition increased time to peak and attenuated moderate floods.

Examples of river morphology restoration with no FCERM monitoring

- **Rottal Burn, Angus.** Approximately 1.2km of the channelised Rottal Burn was restored to its natural meandering morphology. The works cost ~£200,000.
- River Rother, West Sussex. The river was reconnected to a remnant meander (about 850m in length) at a cost of £90,000 (RRC 2016). FCERM was not the main driver for the work.
- **River Ehen, Cumbria.** The Ben Gill, a tributary of the River Ehen, was reconnected to restore the continuity of sediment supply and natural water flows. FCERM was not the main driver of this work.

2.2.6 Funding

There are a number of examples of the use of funding partnerships, whereby different funding sources, some of which may even have been individually justified on non-flood risk management grounds, are drawn together to deliver an integrated project or scheme which delivers multiple benefits. Other potential funding opportunities include:

- Heritage Lottery Fund
- local authority capital grants
- Flood Defence Grant-in-Aid or Local Levy
- agri-environment schemes
- biodiversity offset schemes
- landfill tax credit schemes
- charitable funds

Voluntary and in-kind work should also be encouraged.

The Environment Agency has developed guidelines on the whole life costing of a FCERM project (Environment Agency 2015a), accompanied by a suite of notes covering all aspects of FCERM.¹ However, the report acknowledges there is limited information on whole life costing for a WWNP based project (specifically river restoration) and hopefully further information may come from information presented in the case studies above.

A report commissioned by the Scottish Government assesses the mechanisms for compensating land managers (RPA et al. 2015). The aim was to investigate the options for compensating land managers who implement NFM measures on their land and the report identifies a range of mechanisms by which public bodies can compensate land managers, including land purchase/sale, leaseback and wayleaves. The report also presents a 5-step plan for developing a payment rate for compensation.

2.2.7 Design, management and maintenance

A large number of approaches to river restoration and rehabilitation are available (Federal Interagency Stream Restoration Working Group 2001, Roni and Beechie 2012, RRC 2016).

Selection of the appropriate technique requires an understanding of the processes that underpin channel morphology to ensure that the actions undertaken are appropriate for the particular setting (Kondolf 1998). Hydrological, geomorphological and ecological processes and the human pressures and interventions influencing the reach need to be considered. Understanding of the processes and interventions that have influenced the trajectory of channel and floodplain adjustments which have occurred, the type of river and related river floodplain processes that are to be restored, and societal constraints on the approaches that can be adopted is crucial (Surian et al. 2009, Beechie et al. 2010).

A process-based approach needs to extend beyond the reach that is to be restored to capture processes, pressures and responses operating at the catchment scale. It also needs to extend back in time in order to understand how the reach and its floodplain have responded to changes in natural processes and human pressures, and have been affected by human interventions over recent decades (Arnaud-Fassetta et al. 2009, England and Gurnell 2016). Such an analysis ensures that the causes of degradation are recognised and addressed at the correct scale, using appropriate restoration measures to minimise the likelihood of undesirable responses such as siltation or bed incision at the restoration site. It also helps to identify and tackle pressures that may override any benefits accrued from a restoration (Palmer et al. 2010).

Planning river restoration projects also requires an understanding of societal constraints and desires. The multiple aims of river restoration projects means a trade-off in the selection of river restoration goals or techniques may be required to reconcile any differences to ensure that the best outcome is possible (Dufour and Piégay 2009). Planning river restoration projects also requires an assessment of potential risks and uncertainty to mitigate potential problems after a project has been implemented (Thorne et al. 2015, Erwin et al. 2016). Scores can be attached to the different expected social benefits and ecosystem services created by restoration which may assist planning to maximise these aspects for society (Gilvear et al. 2013).

Ideally, maintenance and management actions should be kept to minimum after restoration actions have been performed. Natural processes should be left to reshape channels to the prevailing flow, sediment and vegetation regimes that are appropriate for a given setting (Kondolf et al. 2006). However, there may be occasions where careful sediment management is required if channel changes are creating undesired geomorphic or flood risk hazards that threaten property and infrastructure. Adaptive management approaches are

¹ Project SC080039. See <u>http://evidence.environment-agency.gov.uk/fcerm/en/Default/FCRM.aspx</u>

needed to monitor responses to restoration actions and implement sensible mitigation actions if required (Downs and Kondolf 2002).

As part of the design process for any river restoration project, a robust monitoring framework should be devised to evaluate the responses and benefits to help inform successful restoration in the future. The RRC PRAGMO document (RRC 2012) and 'Stream and Watershed Restoration: A Guide to Restoring Riverine Processes and Habitats' (Roni and Beechie 2012) provide useful information for devising monitoring strategies. Currently however, there is little guidance on how to monitor the FCERM benefits of river restoration schemes (see Section 2.8).

2.2.8 Research gaps

Evidence of the geomorphic, hydraulic and ecological effects of river restoration has increased in recent years (see Section 2.3), although there is still a need for further monitoring of these aspects. At present, the quantity of field-based empirical evidence of the flood attenuation and FCERM benefits of river restoration is very limited (see Section 2.4). A larger body of direct field-based evidence (alongside model-based understanding) is required to improve confidence. There is a need especially to:

- measure empirically the conveyance capacity of restored rivers compared with degraded ones
- measure empirically the attenuation and water storage effects of restoring natural river processes and landforms
- assess the types, spatial and temporal scales of restoration needed to cause an effect at different locations in a catchment

2.3 Floodplain and floodplain wetland restoration

This section reviews the scientific evidence (including international literature) behind floodplain restoration for FCERM benefits, focusing on the specific measures of floodplain reconnection (including embankment removal and setback) and floodplain wetland restoration.

Floodplains are areas in the bottom of valleys that are constructed by rivers over long periods of time through 2 groups of natural processes. First, during overbank floods, river water spreads out of the river channel across the valley bottom and sediment transported in this water is deposited. These sediment deposits gradually raise the ground surface vertically to form a floodplain. Second, rivers build floodplains laterally as they move across the valley bottom. As the river cuts new channels and erodes its banks, usually on the outside of bends, the eroded sediment is transported and deposited elsewhere, usually within the river channel and most frequently on the inside of bends. This combination of channel and bank erosion and bank construction allows the river channel to move across its floodplain, depositing new areas of floodplain as it moves.

These processes of floodplain construction are naturally linked to processes of river channel development. As a result, distinctive forms of floodplains are associated with distinctive types or styles of river channel (Nanson and Croke 1992). Put simply, floodplains can be defined as areas of the valley bottom that are or were historically inundated periodically by floodwaters (BSI 2005). Several other terms are often used to refer to floodplains or parts of floodplains including alluvial areas, riparian zones, floodplain wetlands and hydraulic floodplains. In addition, the term 'washland' is occasionally used, although it usually refers to engineered zones on floodplains that are surrounded by bunds and have spillways, dams

and floodgates to control flooding on a smaller scale (see Section 2.5 on offline storage areas).

Over the centuries these natural floodplain processes have become increasingly constrained by humans (Beechie et al. 2010), so that many floodplains no longer function naturally or these natural floodplain processes only operate within restricted areas (Lewin 2013). This severe reduction in the natural functioning of floodplains has resulted in reductions in floodwater storage and changes in floodplain morphology (Hudson et al. 2008) and ecology (Lake, 2012).

Human activity over the centuries to control and exploit rivers and floodplains has resulted in only 2% of Europe's rivers and floodplains being classified as natural (Blackwell and Maltby 2006). The extent of disconnection of riparian wetland systems is illustrated by data from the River Habitat Survey (Environment Agency 2003a) which reveal that, of the 24,000 river survey stretches, only 7.1% had a floodplain on either bank for at least a third of the 500m survey length. Of these river stretches with recorded floodplain, 13% had embankments close to the watercourse and 6% had embankments set back. A later survey (Environment Agency 2010) found that 42% of river survey stretches were 'severely modified' with a further 20% 'significantly modified', indicating extensive physical impact on the rivers of the British Isles. In a different analysis at the national scale, comparison of the area of natural floodplain in England and Wales(using the Institute of Hydrology's 100-year flood extent map) with the current area of floodplain (from the Environment Agency's National Flood Risk Assessment dataset) also indicated that 42% of former floodplain has been lost (Maltby et al. 2013).

Floodplain restoration aims to re-establish more natural processes of floodplain inundation, water storage and flood hydrograph attenuation and may be carried out for flood risk management, Water Framework Directive and ecological benefits (including in some cases aesthetic/societal purposes). Floodplain restoration usually forms part of wider river restoration projects (Opperman et al. 2010), but this is rarely the case for washland creation (RSPB 2008). Figure 2.5 illustrates the natural functioning of a floodplain.

The main flood risk management benefit of floodplain restoration is an increased capacity to store water both on and within the floodplain (Hein et al. 2016). This increased storage capacity depends on the area flooded and factors such as the form of the floodplain surface (Lewin and Ashworth 2014), soil permeability, soil moisture retention capacity and vegetation cover (RSPB 2008), all of which are influenced by land use and management (Rak et al. 2016). However, storage capacity is not the only factor that is relevant to FCERM: the roughness of the floodplain surface influences the rate at which water flows over the floodplain. Both storage and roughness attenuate the flood peak, changing the size of peak flows, the shape of the flood wave and the rate at which it propagates downstream (Rak et al. 2016).



Figure 2.5 Illustration of a floodplain from the NWRM.eu website

2.3.1 Understanding the science

The hydraulic roughness or flow resistance of river channels and floodplains can slow the downstream progression of a flood, attenuate its shape and reduce peak flows. Hydraulic roughness is traditionally measured by the well-known Manning's roughness coefficient (n) (Manning 1891). In-channel vegetation, the size distribution of sediment particles, the presence of bedforms and the presence of large wood can increase the roughness of the wetted perimeter of a channel and therefore the Manning's n number. However, more recent research has questioned whether Manning's n is an appropriate index of roughness (Rameshwaran et al. 2011).

When the water level within a river channel exceeds the level of the bank tops, water spills onto the floodplain and therefore the wetted area and roughness increases immensely. In some cases, for instance, it can increase 10-fold (for example, a 1m wide river may have a 100m wide floodplain). Dense, coarse and woody vegetation on floodplains creates higher hydraulic roughness and in turn potentially creates a larger attenuation of a flood peak. Installation of channel embankments generally increases river channel capacity and reduces flow resistance, leading to larger flow velocities and increased discharges within the enlarged channel coupled with a local reduction in floodplain inundation but potentially increased inundation downstream (Gilvear 1999, Darby and Simon 1999, Clilverd et al. 2016).

Coupled with roughness properties (Świątek et al. 2008, Sun et al. 2010), the storage of water on a floodplain is a vital factor in mitigating flooding downstream. Owing to their flat topography and large areas, floodplains can store vast amounts of water. At Mayes Brook in Mayesbrook Park in north London, for example, the restoration of 500m of river along with associated landscaping work created 15,800m³ of extra floodplain storage. Flooding of a 35km² floodplain in the Shannon Valley in Ireland – with an average depth of 1m – equates to a storage equivalent to one day of the peak discharge (around 400m³s⁻¹).

In addition, the restoration of natural floodplain depressions, such as the reinstatement of wetlands, lakes and side channels, increases floodplain roughness, providing additional surface floodwater storage and areas where the retained water can infiltrate into the floodplain to be stored as groundwater.

Floodplain wetlands are an important aspect of floodplains with respect to FCERM and other ecosystem services. Floodplain wetlands serve as nature's method of flood control owing to their short- and long-term water storage capacity (Mitsch and Gosselink 2007, Kadykalo and Findlay 2016). There is evidence suggesting that floodplain wetlands may reduce the frequency (Hillman 1998, Acreman et al. 2003, Acreman and Holden 2013) and magnitude (Ogawa and Male 1986, Ferrari et al. 1999) of flood events and increase the lag time of flood events (Walton et al. 1996, Hardy et al. 2000).

The conservation and/or restoration of floodplains have been promoted for flood risk management on many large rivers of the world, such as the Rhine (Baptist et al. 2004). The functions related to the hydrological processes of wetlands² are some of the most frequently and intensively studied (Bullock and Acreman 2003, Lindsay et al. 2004, Wu and Johnston 2008, Heimann and Krempa 2011, Acreman and Holden 2013). One of the key papers indicating the functioning of floodplain wetlands for reducing floods is the work by Acreman and Holden (2013). Their review found that, in general, floodplain wetlands are better at attenuating flood flows compared with upland wetlands. However, fundamentally, landscape location and configuration, soil characteristics, topography, soil moisture status and management all influence whether these wetlands provide flood reduction services (Acreman and Holden 2013). For example, when saturated, floodplains can become flood generating or 'contributing' areas in some cases (see, for example, Betson 1964, Hewlett

² This statement captures all wetlands, including non-floodplain wetlands.

and Hibbert 1967). If a floodplain wetland is poorly connected with a river, then it is likely to have little impact on downstream flows compared with a wetland hydrologically connected to a river channel.

Internationally, floodplain restoration is commonly used as a FCERM strategy. For example, a river and floodplain restoration project in northern California utilised a 'pond and plug' method to restore the river channel and to restore floodplain wetlands (3.6km of river and 230ha of mountain meadow) (Hammersmark et al. 2008). The study found:

- increased groundwater levels and volume of subsurface storage
- an increase in the amount of times the floodplain was inundated and an associated decrease in the magnitude of the flood peak
- · decreased annual run-off and duration of baseflows

Because groundwater and surface water interactions are linked, it is important to consider groundwater when managing floodplain wetlands, especially when these wetlands have a permeable geology beneath them (House et al. 2016). However, floodplain groundwater systems are complex and a study conducted in Findhorn, Scotland, highlighted the importance of considering groundwater processes in flood management schemes (MacDonald et al. 2014).

2.3.2 Confidence in the science

The science and guidance surrounding floodplain restoration has been growing since the beginning of the century. For example, the European Commission has produced guidelines on how to manage floodplains for flood risk reduction (Blackwell and Maltby 2006). This guidance was informed by literature, workshops and conference events (125 delegates from 19 countries).

Although this report gives an overview of floodplain reconnection projects in Europe, it does not discuss the hydrological benefits in detail.

Nearly all of the research investigating the FCERM benefits of floodplain restoration is based solely on modelling. Empirical studies that assess FCERM benefits are rare). Modelling has been used in some way in most research papers, and there are some examples (see later examples such as Hunworth Meadow) whereby empirical evidence has been utilised in modelling approaches.

Hydraulic models are the most common method for assessing floodplain restoration. Wetland processes are directly incorporated into some modelling packages, whereas these processes have been incorporated indirectly in other modelling approaches (Rahman et al. 2016) such as:

- SWAT (Arnold et al. 1993, Rahman et al. 2016)
- SWIM (Krysanova et al. 2005, Hattermann et al. 2008)
- WATFLOOD (Kouwen 2013)
- MIKE SHE (DHI 2009)
- MODFLOW (Restrepo et al. 1998)
- FLATWOODS (Sun et al. 1998)
- SLURP (Kite 2001)

Results from modelling studies have indicated floodplain restoration could offer positive FCERM benefits. However, Acreman et al. (2003) highlighted a few deficiencies on using hydraulic models to assess the impact of floodplain restoration based on their experience of using the ISIS³ model. These are:

- Use of scenarios: Caution is needed as scenarios are simplifications of what may occur in reality. For example, embankments are never truly level and may have low points which breach first. A model may not capture this process. Therefore, restoration works will need to work within these limits of the scenarios.
- Hydraulic models rarely capture all the losses from a floodplain such as evaporation and infiltration losses. These losses could be captured in the outflow component of a model but this does not represent reality.
- Some models (such as ISIS) do not capture sedimentation and erosion rates (which can affect the hydraulic conditions). Therefore, it is important to consider sediment transport and geomorphology.
- Roughness: This is a key parameter in a hydraulic model, but it can vary significantly (especially on a floodplain). The use of a three-dimensional (3D) computational fluid dynamic model is recommended, but approaches to determine roughness in the field are time consuming, imprecise and costly. Also roughness can change with discharge (Myers et al. 2001). Since the paper by Acreman et al. (2003) was published, however, new technologies such as use of an unmanned aerial vehicle (UAV) and laser scanning surveys (Vaaja et al. 2011) have emerged that could help to resolve these issues. Although hydraulic models are now available to model floodplain flows, the challenge is the appropriate representation of roughness. The interaction between flow and vegetation and physical blockage causes the formation of eddies and wakes in and around plants from the leaf and stem scale to the patch scale. These increase energy losses within the channel and, as a result, increase flow depth. This in turn decreases in-stream velocity. In such cases, flow resistance is derived primarily from the drag force on the vegetation rather than the bed roughness (as represented by Manning's n). Due to the multiscale property of flows within a gravel-bed boundary and complex vegetation structure, the issue of spatial averaging needs to be addressed in any hydraulic models (Nikora 2010). The model is based on the double-averaging methodology (that is, double-averaged continuity and Navier-Stokes equations), which includes drag terms, form-induced momentum fluxes, blockage (porosity) and turbulence effects due to the gravel river bed and in-stream vegetation can be used in 3D modelling (Rameshwaran and Naden 2012, Rameshwaran et al. 2014). Simplified two-dimensional (2D) modelling can be found in Rameshwaran and Shiono (2007) and Sun et al. (2010).

Understanding the extent and characteristics of a floodplain and obtaining more detailed site-specific information remain key challenges. Assessing the size and duration of floodplain inundation remains the area of largest uncertainty in characterising floodplains (Tockner and Stanford 2002). Within Europe, significant data gaps still remain on the delineation of floodplains, characterising floodplain land use and the economic benefits delivered by floodplain ecosystems services (EEA 2016). Clilverd et al. (2016) highlighted the complexity of the effects of river restoration on hydraulic and hydrological processes (and their site-specific nature) and the fact that they are often difficult to determine if there is insufficient monitoring pre- and post- restoration. Floodplain restoration is rarely carried out in isolation, and when linked with riparian planting, the effects of restoration on decreasing flood peaks

³ This model is now called 'Flood Modeller'; however, the literature uses the old model name.

become more pronounced (see Section 3.4 on riparian tree planting). Furthermore, if the restoration of floodplain processes is allowed to incorporate minimal management of floodplain sediment and vegetation dynamics, the resulting increase in the complexity of floodplain microtopography, stratigraphy and vegetation development will further promote floodwater storage and infiltration, and the attenuation of floods as they progress downstream.

The scientific literature surrounding floodplain wetlands is more advanced than literature surrounding other floodplain restoration approaches (for example, embankment setback), where such wetlands include floodplain lakes, ponds and other closed wet depressions as well as wetlands developed within abandoned, often shallow, channel networks and active side channels. Most of the evidence suggests that floodplain wetlands can reduce or delay floods, with examples from all regions of the world. Acreman and Holden (2013) concluded that:

'Floodplain wetlands slow flood wave speed and store large quantities of water, primarily on the surface, that flow back into the river later, evaporate or recharge groundwater. Floodplains with rough vegetation (for example, trees and shrubs) have high friction and slow flood wave speed'.

Bullock and Acreman (2003) reviewed 439 statements that referred to wetlands and the role they played within the hydrological cycle. Out of the 28 that referred to floodplain wetlands, 23 showed that they reduce or delay flood peaks. As the functioning of the wetland is site-specific, the paper highlighted the need for a wetland classification system that accounts for wetland function in a certain location. Similarly, Kadykalo and Findlay (2016) investigated 28 studies that considered the flow regulation services provided by wetlands. They found that, on average, wetlands reduce the frequency and magnitude of floods and increase flood return intervals (while maintaining higher low flows). However, they also reported that in the absence of detailed local observations, estimating flow regulation services generally have large uncertainties.

However, as previously stated, flood attenuation responses are site-specific and some case studies have shown no or a negative effect. For example, Bullock and Acreman (2003) identified some literature that suggested floodplain reconnection in some cases could increase downstream flood risk (for example, issues such as flood peak synchronisation when considering whole catchment flood risk). In such cases, improving river channel connectivity to the floodplain could increase the conveyance capacity thereby counteracting the increase in flood storage provided by the floodplain. Therefore all restoration works should undergo a pre-feasibility study. Figure 2.6 shows an example of floodplain reconnection from Mar Lodge in Scotland (see case studies).

Looking at the full spectrum of all wetland types, there is still a debate about the location and size of wetland that is more effective for FCERM purposes. Mitsch and Gosselink (1993) and DeLaney (1995) argued that many small upland wetlands may be more effective for flood attenuation than a single large downstream wetland. This contrasts with the views of Acreman and Holden (2013), which are more likely to be accurate as they conducted a review of many (more recent) publications. This is reinforced by a modelling study on the Charles River, Massachusetts, which suggested that downstream main-stem wetlands (linked with fifth order rivers⁴) were found to be more effective in reducing downstream flooding than upstream wetlands (associated with rivers less than fifth order) (Ogawa and Male 1986).

⁴ Stream ordering is a way of classifying the size of rivers. See <u>http://www.fgmorph.com/fg_4_8.php</u> for more details.



Figure 2.6 Floodplain reconnection through embankment lowering on the upper River Dee near Braemar, Aberdeenshire, October 2015 (© Anke Addy)

It is likely that scientific uncertainties in restoration coupled with land use issues (that is, compensation for the periodic inundation of fields) are factors that have traditionally constrained floodplain restoration (Brookes and Shields 1996, Adams and Perrow 1999). It is challenging to draw specific conclusions on the magnitude of benefits owing to the site-specific nature of floodplain hydrology and topography. Therefore, it is difficult to predict the effect of floodplain reconnection on reducing flood peaks without conducting a site-specific hydraulic modelling study (Jacobs 2011, SEPA and Forestry Commission 2012). In a review of floodplain restoration (incorporating tree planting and wetland creation) on FCERM at smaller scales. The review highlighted the multiple benefits of floodplain restoration of communicating these clearly to stakeholder and landowners from the inception of restoration (Ball 2008).

2.3.3 Metrics of cost–benefit and the multiple benefits

Some of the international literature has attempted to look at cost–benefits (or just costs) in detail through peer-reviewed studies. For example, the Charles River Study, Massachusetts, by the US Army Corps of Engineers concluded that a loss of 40% of the wetlands within the catchment would increase flood damage by at least \$3 million annually (Sather and Smith 1972). It was also estimated that 38km² of floodplain storage on the Charles River saved \$17 million of downstream flood damage each year (Sather and Smith 1972). The study therefore highlighted the need to protect such valuable wetlands. The floodplains of Illinois were valued as high as \$7,500ha per year with 86% of this valuation based on regional floodwater storage (Sheaffer et al. 2002, Tockner and Stanford, 2002).

There could, however, be a conflict between FCERM functions and ecological objectives when a floodplain wetland water table is managed at a high level (thereby reducing flood storage capacity) (Acreman et al. 2007). It should be noted that the examples above are on large-scale floodplain systems in the USA and that the sizes and scales of these floodplains are larger than UK floodplain systems. However, the examples provide a broad working of cost–benefits. Within Europe, Grygoruk et al. (2013) presented an economic assessment tool for the value of floodplain water storage in the Biebrza Valley, Poland. They found through a geographical information system (GIS) based study on hydrological floodplain processes (over a 10-year period) that the average annual volume of active water stored on the floodplain reached ~10 million m³ per year, equating to a monetary value of €5.5 million per year.

There is an abundance of literature highlighting the multiple benefits for floodplain restoration. Floodplains are generally classified as biogeochemical hot spots for carbon and nitrogen cycling and storage (McClain et al. 2003, Welti et al. 2012). Depending on the frequency of inundation and other site conditions (for example, vegetation), they can become either carbon and/or nitrogen sinks or sources (Pinay et al. 2007, Welti et al. 2012). Welti et al. (2012) concluded that, by increasing the frequency of flooding into floodplain backwaters, nitrous oxide production could be mitigated and the nitrate removal capacity of a floodplain could be increased. They demonstrated this using a case study site on the Danube where creating regular surface water connections to the river could reduce nitrous oxide emissions by 50% at a site that is currently disconnected.

Floodplain wetlands can reduce nitrate loads, Mitsch and Day (2006) calculated that 2.2 million ha (0.7% of the catchment area) of wetlands are needed in the Mississippi–Ohio–Missouri (MOM) river basin to sufficiently reduce the nitrogen loads to the Gulf of Mexico to ensure a reduction in the size of the hypoxia. The authors noted that this restoration would offer other benefits such as flood reduction, water quality improvements, reduction in health threats and new habitats, though none of these benefits was quantified. The MOM study suggests that large-scale scientific research is needed to reduce the major uncertainties surrounding catchment-scale benefits (Mitsch and Day 2006). Inundation of floodplains may result in nutrient enrichment of soils from flood-deposited sediments and nutrient-rich water (Gowing et al. 2002, Clilverd et al. 2013). However, Koontz et al. (2014) found that measurements of total carbon, total nitrogen and phosphorus were all reduced at restored floodplain wetland sites on the lower Mississippi compared with reference sites.

Natural floodplains are among the most biologically productive and diverse ecosystems on the planet (Tockner and Stanford 2002). Floodplain diversity and productivity can be linked to dynamic and variable connectivity with river flows, that is, the periodic inundation through flooding is most of the time responsible for high floodplain productivity (Junk et al. 1989). High energy flows across floodplains and through river channels induce erosion and deposition, creating complex floodplain microtopography and stratigraphy which support high habitat heterogeneity and high levels of biodiversity (Salo et al. 1986, Opperman et al. 2010).

Monitoring and measurements to quantify floodplain restoration success are rarely implemented, even though this information would be useful to guide other restoration projects and assess the multiple benefits of restorations (Buijse et al. 2002, Downs and Kondolf, 2002, Palmer et al. 2007, Lamouroux et al. 2015, Riquier et al. 2015). Riquier et al. (2015) and Hudson et al. (2012) noted the need for further research to develop methods and indicators to assess floodplain habitat conditions.

Floodplain restoration that achieves flood attenuation and promotes water quality improvements and increased biodiversity does not need to result in the loss of another ecosystem services. However, it is inevitable that there will be some trade-offs in restoration (see Acreman et al. 2011). If the primary purpose is to lower flood risk downstream, then the best approach to flood plain restoration is to maximise the storage area, regulate inflow and out flows (see offline storage areas review in Section 2.5). Opperman et al. (2009) proposed a vision whereby we do not empty floodplains of human activity but an approach that would reduce unsustainable uses while maximising floodplain benefits for both society and private landowners. They illustrated this view by citing numerous examples from across the USA, including the Yolo Bypass – a large-scale floodplain reconnection project connecting over 240km² of floodplains.

2.3.4 Effectiveness/performance

The most effective floodplain reconnection projects are those which take place on wide floodplains with little risk to nearby infrastructure (SEPA 2016). However, not all floodplains

are flat. For example, undulating, rough and complex floodplains such as upland floodplains can provide large amounts of water storage and therefore attenuation capacity. For example, on the River Waal in the Netherlands, the Cyclic Floodplain Rejuvenation project was able to sustain safe flood levels when ~15% of the total floodplain was restored (Baptist et al. 2004). Williams et al. (2012) reported that wetlands may readily attenuate 'hydrological' floods (that is, high frequency, low return period floods), but are much less likely to attenuate 'economic' floods (that is, low frequency, high return period events resulting in economic damage). As flood height may be a critical determinant of the economic cost of a flood, the management of floodplains upstream of sensitive areas to maximise flood storage and thus to reduce flood height (although not duration) is likely to be the most cost-effective use of wetlands for flood attenuation (Williams et al. 2012).

A UK case study (Hunworth Meadow) assessed the impact of a floodplain restoration scheme for FCERM (Clilverd et al. 2016), Hunworth Meadow is a 400m long, 40-80m wide (3ha) section of floodplain beside the River Glaven in Norfolk. The catchment area above this point on the river is 115km². Until 2009, the river was disconnected from the floodplain by an embankment. However, in the spring of 2009, the River Glaven Conservation Group, the Wild Trout Trust and Natural England removed the embankment. The study by Clilverd et al. (2016) is rare as it uses an empirically driven hydrological/hydraulic modelling approach and contains data from before and after restoration. Not many examples of such detailed before-after studies exist in the UK. MIKE SHE/MIKE 11 was utilised to analyse the impact of the floodplain reconnection before and after restoration. Using data from 2007 to 2010, the study found that the removal of the embankment created inundation of the floodplain at high flows (>1.7m³s⁻¹). However, the restoration resulted in only a small impact on flood peak attenuation (maximum 5% peak reduction) owing to the limited length of restoration and improvements in the drainage back into the river (Clilverd et al. 2015). Clilverd et al. (2013) found that the removal of embankments reduced the channel capacity by ~60% and suggested overbank flooding was the most dramatic hydrological effect following the restoration of the site, although groundwater levels were also higher and subsurface storage was greater. However, Acreman et al. (2011) concluded that, in most floodplains, soil water storage usually plays a minor role compared with surface topography, and that storage in floodplain ditches may have a small influence in comparison with surface storage in other areas of the floodplain. Nevertheless, it is important to consider groundwater when planning a floodplain restoration scheme.

In the River Cherwell catchment (a tributary of the River Thames, catchment area of 910km²), Oxfordshire, a modelling study was conducted to assess the impact of floodplain reconnection works (Acreman et al. 2003). The study's objectives were to assess: the impact of the proposed restoration; and the increased inundation of the floodplain itself as a means of rehabilitating wetlands. However, the study only used hypothetical scenarios for restoration as no restoration works were undertaken nor planned. The research used a one-dimensional (1D) hydraulic model coupled to a semi-distributed hydrological model. The study highlighted that representing the complexities of floodplain hydraulics in the model proved challenging (increased uncertainties) (Acreman et al. 2003). However, the results were promising, suggesting that:

- where embankments were in place, the peak discharge could increase by 50– 150% (using a scenario of adding in embankments to areas where embankments had been removed)
- reconnecting the river channel to the floodplain (pre-human intervention scenario) could increase peak water levels on the floodplain by 0.5–1.6m (thereby reducing the flood peak by around 10–15%)

However, Acreman et al. (2003) also suggested that not all floodplain reconnections have such a substantial impact and that it might be more representative to anticipate a 0-15%

reduction in peak flow when entrenched channels are reconnected to floodplains. Thus removing embankments may have limited impact if the river has been dredged and the floodplain is inundated only at high return periods. The Cherwell study tried to address this by examining the combined impact of channel (reinstating natural cross-sectional geometry) and floodplain (embankment construction/removal) restoration. The modelling showed that in one scenario where embankments were introduced, the time of peak flow could be brought forward by 40 hours, whereas in a natural conditions scenario, the peak could be delayed by ~17 hours (Figure 2.7).



Figure 2.7 Impact of floodplain restoration at Easter 1998 at Clifton Hampden (River Thames, Oxfordshire)

Source: Acreman et al. (2003)

The difference between the study by Clilverd et al. (2016) at Hunworth Meadow and Acreman et al. (2003) on the River Cherwell is that the latter used a scenario to simulate the effect of flood embankment removal while the former used a scenario to reintroduce the embankments. Furthermore, the study by Clilverd et al. (2016) was driven by data pre- and post- restoration while Acreman et al. (2003) could only employ pre- restoration data. However, both studies offer valuable knowledge to the UK database on floodplain reconnection studies.

A modelling study of restoration scenarios was conducted for the Thur River catchment, a mountainous area in north-east France of 260km² (Kreis et al. 2005). The catchment has a flashy response owing to its steep topography. The study noted the need for high resolution topography data in order to run the model correctly. A 1D hydraulic model was utilised (HEC-RAS) and the study found that re-naturalising the channel and floodplain did not reduce the flood peak. The authors believed this to be a result of the bed being partly incised. However, recent studies such as SEPA and the Forestry Commission (2012) have shown limitations with modelling approaches such as those utilised by Kreis et al. (2005) (for example, 1D hydraulic modelling – see Section 3.2).

2.3.5 Key case studies

There is a growing number of case studies internationally that provide evidence for the impacts of floodplain restoration. For example, in Germany, the saying 'mehr Raum fur Flusse' (more space for rivers) has been used since the 2002 floods (Kundzewicz and Menzel 2005). An example case study of this strategy comes from near Lenzen in Brandenburg, Germany, where embankments have been set back and a floodplain area of 300ha created (DKKV 2004).

Many examples have already been described above. However, the majority of floodplain restoration sites are unmonitored. In addition, floodplain restoration is usually conducted at
the same time as channel restoration, making it difficult to separate the impact of the 2 types of restoration.

Below are some examples of established case studies of monitored and unmonitored catchments in England:⁵

- Hunworth Meadow, River Glaven, north Norfolk (summary of Clilverd et al. 2016). Hunworth Meadow is a 400m long, 40–80m wide (3ha) section of floodplain beside the River Glaven. The catchment area above this point on the river is 115km². A summary of the scientific research in this monitored case study is given above.
- **Mar Lodge**, River Dee, Aberdeenshire (summarised in Addy et al. 2016). An artificial embankment was partially removed in October 2015 to reconnect part of the floodplain (upstream catchment area of 370km²). In summer 2013, monitoring of the floodplain subsurface water table (and surface inundation levels during a flood) commenced, giving 2 years of valuable background data. River discharge data are available from the Mar Lodge gauging station. The project has one year of post-restoration data including data from Storm Frank. Although further data are required in order to assess the hydrological changes at the site, there have been significant geomorphic changes around the restoration works post-flood.
- **Rivers Sow and Penk**, Staffordshire (from Jones 2010). Floodplain restoration work, with some basic monitoring, was carried as part of the Farming Floodplains for the Future project. The main measures were washlands (see later review), but the project also restored ~100ha of floodplain meadows.
- Over 500 metres of **Mayes Brook**, north London was restored and 15,000m³ of new floodplain storage was created (London Borough of Barking and Dagenham 2012, Driver 2016). This type of floodplain lowering may need long-term management as the river re-deposits sediments, with enhanced rates of sedimentation achieved around colonising vegetation. However, owing to the size of this river, this is unlikely to be an issue at this case study site. In the Netherlands, 'cyclic floodplain rejuvenation' is used to deal with this issue (Baptist et al. 2004).
- Another example case study from north London is the **River Quaggy** restoration works. Driver (2016) highlighted that the restoration and creation of floodplain wetlands in this urban catchment created had 85,000m³ of floodplain storage space and significantly reduced the flood risk to 600 properties.

2.3.6 Funding

The cost of floodplain restoration works depends on the scale and the complexity of the project and should include a pre-feasibility study (SEPA 2016). There are a number of examples of the use of funding partnerships, whereby different funding sources, some of which may even have been individually justified on non-flood risk management grounds, are drawn together to deliver an integrated project or scheme which delivers multiple benefits.

Potential funding opportunities include:

• Heritage Lottery Fund

⁵ This section gives a summary of these studies. More details of the science behind some of them are given in previous sections.

- local authority capital grants
- Flood Defence Grant-in-Aid or Local Levy
- agri-environment schemes
- biodiversity offset schemes
- landfill tax credit schemes
- charitable funds

The Environment Agency has developed guidelines on the whole life costing of a FCERM project (Environment Agency 2015a), accompanied by a suite of notes covering all aspects of FCERM.⁶ However, the report acknowledges there is limited information on whole life costing for a WWNP based project (specifically floodplain restoration) and hopefully further information may come from information presented in the case studies above.

A report commissioned by the Scottish Government assesses the mechanisms for compensating land managers (RPA et al. 2015). The aim was to investigate the options for compensating land managers who implement NFM measures on their land and the report identifies a range of mechanisms by which public bodies can compensate land managers, including land purchase/sale, leaseback and wayleaves. The report also presents a 5-step plan for developing a payment rate for compensation.

2.3.7 Design, management and maintenance

It is essential to carry out pre-works assessments and surveys to be sure that the restoration does not increase flood risk (especially locally if receptors are present) and cause negative affects to the local ecology (SEPA 2016). For example, the management of ditches and water table levels on floodplains can influence flood attenuation processes on floodplains.

Acreman et al. (2011) investigated the trade-offs between flood management and ecological benefits on the Somerset Levels. It was calculated that increasing the water levels in ditches in one case study catchment to meet ecological targets over winter would result in a loss of overall flood storage capacity (which equates to 2% of the medium annual flood volume) (Stratford et al. 2015). There is also a risk that some floodplain reconnection projects that reduce channel entrenchment could result in increased water levels. Under these circumstances, the backwater effect should be investigated (SEPA and Forestry Commission 2012).

One option for floodplain restoration where embankments are present is to Do Nothing (through natural recovery), that is, to allow nature to naturally break down an embankment over time at the desired location. Of course, there is no answer as to when nature will eventually break through an embankment, and so controlled removal is often preferred. However, over an 8-year period, 228 breaches of flood embankments were recorded on the River Tay in Scotland (owing to a high number of high discharge flood events reaching return periods of 120 years) (Gilvear and Black 1999), with overtopping identified as the main mechanism that resulted in embankment failure.

The slopes of the river and floodplain have crucial influences on the retention potential of a floodplain. Habersack et al. (2008) stated that:

⁶ Project SC080039. See <u>http://evidence.environment-agency.gov.uk/fcerm/en/Default/FCRM.aspx</u>

'shallow slopes reduce discharge peaks and prolong the retention periods, while steeper slopes worsen the effect of retention, especially when the flood wave is totally discharged in the channel'.

Land acquisition may be required to conduct large-scale restoration projects. According to the European Commission's Natural Water Retention Measures (NWRM) initiative,⁷ the maintenance costs are generally 0.5–1.5% of the investment cost.

2.3.8 Research gaps

In relation to the management of flooding there are a number of knowledge gaps surrounding floodplain restoration. The literature suggests there are a number of research gaps that still need to be addressed to fully understand how floodplain restoration alters hydrological processes during floods. There are other gaps, for example, around management issues. Known gaps are summarised below.

- Further empirical observations of the effectiveness of floodplain restoration for flood risk management are required. Coupled with this, empirical observation of flood regulation services of wetlands is needed.
- Further research is required to understand floodplain functioning during extreme events and the degree to which large-scale floodplain restoration mitigates flood peaks.
- Data gaps still remain on the delineation of floodplains.
- For catchment-scale floodplain restoration, further information on the phasing of subcatchment flows and catchment-scale benefits is required. There is also a need to examine the issues relating to the synchronisation of flood peaks surrounding large-scale floodplain restoration works.
- There are issues surrounding floodplain roughness, for example, the parameterisation of drag coefficients requires further work.
- How can we better understanding complex floodplain hydraulics and represent them in models?
- The work of Clilverd and colleagues has suggested further work is required on the role of groundwater in floodplain restoration (in multiple settings).
- The relative effectiveness of different types of wetland with respect to flow regulation should be assessed.

2.4 Leaky barriers

Leaky barriers are mainly composed of pieces of wood, occasionally combined with some living vegetation, that accumulate in river channels as well as on river banks and floodplains. Although the word 'barrier' evokes thoughts of hard engineering, leaky barriers occur naturally along rivers as a result of:

- local (mainly tree) fall
- water transport and snagging of twigs, branches, logs and trunks
- the construction of lodges and dams by beavers

⁷ <u>http://nwrm.eu</u>

Similar structures can also be engineered by humans.

Leaky barriers are known by many other names. In scientific studies, they are referred to as coarse woody debris (old term) or large wood (current term) pieces, accumulations, jams and dams, and where specifically constructed by beavers, beaver dams or lodges. When engineered, they are often referred to as wood placements, engineered log jams or flow restrictors.⁸ However, the newly introduced term 'leaky barriers' refers to the general functioning of all of these naturally occurring or engineered wood features plus other leaky measures such as boulder placements. These are all examples of natural or natural-looking 'leaky barriers'.

Since these largely woody features are a natural consequence of the presence of riparian and floodplain woodland, one option to encourage their formation is to promote the natural recovery or to plant such woodland in order to reinstate the natural wood cycle (Collins et al. 2012), since mature and senescent trees naturally deliver wood to river systems (see Figure 2.8 and Section 3.4). But because this process takes time, it is useful to speed it up by introducing engineered leaky barriers. There are many examples of WWNP schemes that are engineering leaky barriers within channels to mimic naturally occurring wood while riparian planting establishes within the catchment (Dodd et al. 2016, SEPA 2016, Environment Agency, undated a).



Figure 2.8 A natural in-stream wood accumulation caused by the collapse of a tree in the Logie Burn catchment, Aberdeenshire (© Steve Addy)

Although wood, particularly if it is large in relation to the channel width and is located in naturally functioning and complex woodland river channels, is not particularly mobile (Gurnell and Sweet 1998, Bertoldi et al. 2013), the advantage of engineered leaky barriers is that they can be introduced where they may be most effective for FCERM and they can be fixed in place, thereby reducing risks of mobilisation and blockage of bridges and culverts downstream. However, engineering a stream barrier to mimic natural processes involves understanding those natural processes as well as working closely with all the relevant regulative bodies, since placing wood in a stream incorrectly could induce wood mobility problems and impact on freshwater ecology (Dodd et al. 2016). There may sometimes be a case to remove wood from river channels in some places (SEPA 2016), for example, near sensitive bridges and culverts which, if blocked, could create significant flooding

⁸ This review may refer to these terms when this was the terminology used within the source.

consequences. There are also measures, such as wood racks, that can help to trap mobile wood before it reaches such sensitive structures (Ruiz-Villanueva et al. 2016).

This review examines first at the scientific evidence behind both naturally occurring wood accumulations and those which are engineered by humans. However, this section looks predominantly at the effectiveness and management of those leaky barriers that are installed by humans. It is assumed that future naturally occurring woody debris will result from riparian planting and features that are installed now are intended to have an instantaneous impact on flood peak reduction.

2.4.1 Understanding the science

This subsection considers naturally and engineered wood in rivers, and beaver dams. It begins by reviewing the scientific literature and evidence on both natural and engineered leaky barriers and their role in FCERM. When considering leaky barriers in channels for FCERM purposes, they are likely to be engineered while following the scientific principles of naturally occurring wood as best as possible. Therefore planning, maintenance and management will be based around engineered wood in rivers. Where riparian planting is considered, naturally occurring wood is likely to occur in the future. The section ends by discussing beaver dams.

Leaky barriers vary widely in their size and structure, and the degree to which they extend into and across river channels. However, their overall effect is to retain and slow the movement of water, attenuating flood waves and reducing average flow velocity. Water retention occurs immediately upstream of the wood feature and in side channels and floodplains onto which water may be diverted during high flow events. This storage and attenuation of flows helps to increase both lateral (floodplain) connectivity and vertical connectivity with groundwater through the river bed. At the same time, the complex hydraulic effects of leaky barriers increases channel and floodplain sediment retention, the attenuation of downstream sediment transport, the sorting of river bed sediments, and the creation of a complex of associated bed, bank and floodplain forms, all of which have important ecological effects. The positioning of engineered leaky barriers in appropriate locations can increase local attenuation of floods, increase in-channel water storage and deflect high flows onto desired areas of floodplain.

Wood accumulations are a natural feature of rivers draining wooded river corridors. Under unmanaged conditions, it has been estimated that around 100m³ of wood are likely to be retained within every hectare of river channels draining what would be the natural vegetation cover of most floodplains in the UK – deciduous or mixed woodland (Ruiz-Villanueva et al. 2016). Natural large wood accumulations in rivers contribute to increasing the hydraulic roughness of river channels, influencing flow retention and velocity at both the reach and local scale (Hygelund and Manga 2003). However, it is important to realise that natural wood accumulations vary greatly in their size and form, and that their frequency and morphology change downstream along rivers and according to the style of river and the tree species from which the wood originates (Abbe and Montgomery 2003, Gurnell 2013).

In a survey of 50 restoration projects in Germany and Austria, Kail et al. (2007) found that wood measures are most successful if they mimic natural wood accumulations. In the same study, fixed wood structures were classified as hard engineering, and non-fixed (soft engineered) leaky barriers resulted in the formation of more natural channel features and were also more cost-effective. Unfortunately, the potential flood control benefit of large wood accumulations has been lost in many countries due to the clearance of riparian woodland for farming and the removal of wood from rivers by fishery groups and others (Thomas and Nisbet 2012).

In the UK, the New Forest provides long-term research results concerning the frequency and dynamics of wood accumulations and their association with flow attenuation and channel morphology. Wood in New Forest river channels draining the open forest is largely unmanaged, whereas both wood and river channels have been subject to management within enclosures containing tree plantations. Gregory et al. (1993) observed an average of 1.15 wood jams per 100m along 22km of river channel in the New Forest, with a maximum of 4.3 wood jams in a small stream draining an area of mature, unmanaged, deciduous woodland. In a more detailed, multi-temporal study of a 5.9km length of the main Highland Water, Gurnell and Sweet (1998) observed a maximum spacing of 6.9 wood accumulations per 100m of channel. The spacing of these wood accumulations varied with channel width, typically occurring every 6-9 channel widths. This illustrates the importance of the ratio of wood piece length (dependent on tree size) to the width of the river channel, with spacing being one of many properties of wood accumulations that changes progressively as channel size increases. Furthermore, the lowest frequencies of wood accumulations occurred in a straightened reach draining a coniferous plantation, illustrating the impact of wood management and a simplified channel morphology on wood retention.

Gurnell and Sweet (1998) found that, although the dam type (leakiness and extension across the channel) varied through time, wood accumulations tended to re-form in the same locations following natural wood disturbance or dam clearance. This indicated that, in naturally formed, complex channels with a width less than the typical tree height, large wood pieces are only mobile for short distances before they become snagged. Furthermore, reaches with more frequent wood accumulations were associated with an increase in pool (and associated bar and riffle) formation (as noted previously by Gregory et al. 1994). In a more recent survey of floodplain formation and morphology along the Highland Water, Sear et al. (2010) demonstrated how wood accumulations are also a major influence on side channel formation, with complex channel networks and floodplain morphologies developing quite rapidly after the establishment of major in-channel wood jams. Figure 2.9 shows an example of a flow restrictor during normal flows and a flood.



Figure 2.9 A flow restrictor measure during normal flows (left) and a flood (right) (© Mark Wilkinson)

Notes: The measure is designed to spill water onto a floodplain and store it in an offline pond in the Belford catchment. It is also designed to allow fish passage.

Although based partly on observations from the New Forest, Environment Agency (1999) summarised wider information concerning the presence, nature, dynamics, hydraulic and geomorphological roles of 'large woody debris' in British headwater rivers to propose wood management guidelines. In particular, the report presented field observations of the impact of wood accumulations of different type on flow velocity, flow resistance and the size of aggregated dead zones as flow increases in river channels up to bankfull. These observations illustrate the important impact of wood jams on flow attenuation and storage at

a reach scale in comparison with otherwise similar reaches where no wood is present. Water storage and flow attenuation was greater in reaches with wood accumulations than in control reaches, but the magnitude of the difference became smaller as flow stage approached bankfull.

These impacts on water storage and flow stage suggest that wood placement is potentially useful for raising water levels within incised channels and reconnecting floodplains (Nisbet et al. 2011a). Focusing on floodplain features (that is, using woody brash to roughen floodplains), some studies show that roughening the floodplain through riparian tree planting has a greater attenuation effect than the installation of in-stream leaky barriers (Odoni and Lane 2010, Nisbet et al. 2011a, Dixon et al. 2016). Directly increasing floodplain roughness by planting trees is likely to be the most cost-effective method of enhancing floodplain roughness (see, for example, Sholtes and Doyle 2011). However, trees can take time to establish and so soft engineered approaches such as brash barrier placements can be used to rapidly increase floodplain roughness, while the trees and other vegetation establish and mature.

It is important to note that trees and wood are a part of the same natural cycle. Tree planting or the natural recovery of riparian woodland will not only increase floodplain roughness, but will eventually lead to the release of wood and the natural development of wood jams and other wood accumulations. Furthermore, wood placements rarely achieve the forms and densities of naturally occurring wood accumulations, and so they usually perform very differently. Wood placements are often secured to prevent their movement, but this makes them quite rigid. In comparison, natural wood jams move slightly as water levels change. This allows them to:

- bed down within trapped sediments and become secured in a natural way to the river banks and bed
- act as natural wood traps with the largest, most stable pieces of wood retaining smaller more mobile pieces
- act as truly 'leaky barriers' as the contained wood pieces adjust so that water flows underneath and through them during high flows, but is more strongly retained behind them at low flows

Beavers alter river systems through dam building and other activities. Because beaver dams are more watertight than wood dams, they not only attenuate channel flows but also modify the hydrology of the riparian zone, driving seepage into the banks, bed and riparian zone and releasing water during dry periods (Giriat et al. 2016). Beavers cannot build dams where the flow rate of a channel is too great and this is a reason why beaver dams are more commonly found on smaller channels less than 6m wide (Gaywood 2015).

Beavers are considered 'ecological engineers'. A report by the Beaver Salmonid Working Group (2015) reproduces the following statement by Butler and Malanson (1995):

'more than any other animal except humans, beavers geomorphically alter the landscape through their dam building and related activities'.

The likely physical characteristics and hydraulic, hydrological, sedimentological, geomorphological and ecological functioning of beaver lodges, dams and meadows in the context of the English landscape have been fully reviewed by Gurnell et al. (2009). Because beaver dams are more robustly constructed than wood jams, their hydrologic, hydraulic and sediment retention properties are more marked. Furthermore, when they start to leak, beavers usually rapidly repair them, and when they are abandoned and start to decay, their function is often replaced by other dams located nearby. Gurnell (1998) reported that, although a single beaver pond may have a negligible effect on river flows, a sequence of ponds may have a larger effect. However, the extent to which a beaver pond mitigates flood

risk depends on the available storage behind the dam (Gurnell 1998, Gurnell et al. 2009). There is a risk that large beaver ponds that fail may increase downstream flooding ((Butler and Malanson 2005), although this risk is often reduced by beavers constructing ponds and meadows downstream (Gurnell et al. 2009). This statement is supported by some findings by Hillman (1998), who studied the failure of a large beaver dam in Alberta where the sudden release of 7,500m³ of water produced a flow peak that was 3.5 larger than the maximum recorded discharge. However, it was noted that this peak was reduced to 6% of its estimated upstream magnitude by the flood attenuation of a 90ha wetland containing a small lake and several beaver ponds (Hillman 1998).

2.4.2 Confidence in the science

This section reviews the confidence in the scientific evidence. It begins by addressing naturally occurring wood in rivers before examining engineered leaky barriers and then beaver dams. It looks at some specific studies under each grouping, then finally summarises the confidence in the science as a whole.

Environment Agency (1999) reviewed the impact of large woody debris (LWD) in UK headwater streams. The report concluded that LWD accumulations cause an increase in flow resistance of river channels, which can be considerable at lower flows. The review also found that the resistance caused by LWD leads to an increase in mean flow depth and velocity and increase in floodplain water storage. Although a single LWD may have limited impact on attenuating a flood peak, the aggregate effect may be very significant (Environment Agency 1999). The report suggested that indiscriminate removal of LWD from in headwater rivers should be avoided. Where LWD blockages of man-made structures form a FCERM problem in headwater streams, however, complete removal of LWD is required.

Gregory et al. (1985) found that there was a difference in travel time of over 100 minutes for the situation with and without dams for a discharge of 0.1m³s⁻¹ but a difference of only 10 minutes for a discharge of 1.0m³s⁻¹ along the same 4km channel reach in the New Forest (based on analysing observed hydrograph travel times).

Numerous studies have modelled naturally occurring wood in rivers. For example, Dixon (2013) found that naturally occurring log jams account for 65% of flow resistance in forested river channels; this rose to 75–98% where the log jam was inducing a distinct step in the water profile. However, when modelling log jams alone, Dixon (2013) found a variable response with less clear spatial trends than for forest restoration, and also noted issues with synchronisation of flood peaks.

Focusing on man-made leaky barriers mimicking naturally occurring woody accumulations in watercourses, the majority of scientific evidence in the context of FCERM is based around modelling studies rather than empirically based studies. One example of an empirically driven study is given here. Large wood was installed and secured in the Ore Mountains, south-eastern Germany in a 282m long first order channel. Using an artificial flood wave (for a 3.5-year return period event), results showed a significant delay of the flood wave propagation over the local reach as a result of increased channel roughness (Wenzel et al. 2014). The study also noted a small decrease of 2.2% in peak discharge.

Results from modelling studies are promising. Modelling studies at the reach scale have generally shown that wood placement measures can slow the flood wave locally and delay the timing of a flood peak (see, for example, Kitts 2010, Thomas and Nisbet 2012, Dixon 2013), although these studies have varying results. Nisbet et al. (2011a) reviewed the benefits of leaky barriers in the Pickering catchment and showed that these measures can delay the travel time of the flood peak. In Pickering, Odoni and Lane (2010) utilised the OVERFLOW model and found that the installation of 100 leaky barriers could reduce the magnitude of a flood event by 7.5% (from 29.5m³s⁻¹ to 27.3m³s⁻¹). The study showed that the

magnitude of the reduction increased for larger events when modelling incorporated both riparian planting and leaky barriers. This is slightly different to the observations of Gregory et al. (1985). This difference could be a result of the catchment setting or uncertainties in the modelling assumptions, but it should be noted that the authors acknowledge uncertainties in the modelling approach and highlight some critical assumptions. For example, leaky barriers are represented by changing the Manning's n value and this value is different for a naturally occurring leaky barrier compared with an engineered rigid barrier. Environment Agency (1999) noted that natural, less rigidly secure wood lifts with stage, leading to a lower n value for a particular reach than might be anticipated and a strongly varying n value as stage increases. However, it should be noted that the difference in Manning's n between channels that have LWD and those that do not converges with increasing discharge (Environment Agency 1999). This finding illustrates that there is not a specially designed tool for modelling leaky barriers; most tools take a bespoke approach to existing models using assumptions or features that may not be applicable to leaky barriers. This need for a new leaky barrier tool was highlighted by Environment Agency (2014a). Another example of a modelling approach to leaky barriers is the study by Thomas and Nisbet (2012), which study utilised the Channel Blockage function in the InfoWorks RS model rather than increasing Manning's n (roughness). With a 70% blockage factor applied (to represent a partial blockage of the channel), they found that leaky barriers could increase the travel time of a flood wave over a channel reach by 2-3 minutes. But owing to the uncertainty in flood modelling studies addressing flood peak attenuation that are not informed by empirical data, the confidence in the science for FCERM of leaky barriers is somewhat limited.

Figure 2.10 shows an example of a series of flow restrictors from the Bowmont catchment in the Scottish Borders.



Figure 2.10 A series of flow restrictors located in a small upland gully in the Bowmont catchment in the Scottish Borders(© Steve Addy)

A detailed review on beavers by Gaywood (2015) states that habitat change brought on by beaver activity might contribute to restoring natural processes within catchments. The author suggests that beavers might increase or reduce flood risk at the local level. The report gives inconclusive findings with regards to beaver activity and FCERM objectives, and suggests strategic and local flood risk management planning will need to take account of potential beaver activity with regards to FCERM (Gaywood 2015). The Beaver Salmonid Working Group (2015) suggests that beavers may help with the mediation of flows by stabilising stream flow, and by reducing the size of floods.

There is a growing evidence base to suggest that beaver dam failure does not increase flood risk locally (Campbell-Palmer et al. 2016). However, a conflicting study has shown that

during extreme events there is a risk of beaver dam failure, resulting in extensive disturbance to channel banks and areas of the adjacent riparian area, together with scour and/or aggradation below the washed out dam (Green and Westbrook 2009).

Local scale studies have shown promising results similar to other leaky barrier studies (Gaywood 2015). A temporal analysis of floods in the Ourthe Orientale basin in Belgium showed a reduction in the frequency of major floods since the establishment of beaver dams. For instance, the recurrence interval of a reference flood of 60 m³s⁻¹ increased from 3.4 years to 5.6 years (Nyssen et al. 2011). However, where beaver dams are not maintained there is a risk of loss in water retention. For example, beaver dams in north-east Poland that were not maintained by beavers over a 3–5 year period showed a reduction in water retention from 15,000m³ to 7,000m³ (Grygoruk and Nowak 2014).

Recent results from beaver trials in Devon have shown that the creation of 13 beaver dams has increased water storage within the site by ~1,000 m³. This appears to have caused significant flow attenuation impact, with peak discharges being reduced by $30 \pm 19\%$ and times between peak rainfall and peak discharge increasing by $29 \pm 21\%$. In addition, the beaver ponds have induced a reduction in suspended sediment, nitrogen and phosphate concentrations, which when combined with the attenuated flows, results in lower diffuse pollutant loads downstream. However, dissolved organic carbon concentrations increase (Brazier et al. 2016, Puttock et al. 2017). Although this study gives positive conclusions from the small 30ha catchment, further research is required to understand how the results upscale.

To summarise the above, the scientific literature shows that leaky barriers comprised of wood can generally reduce flood peaks and delay the progression of the flood wave, though if engineered and placed incorrectly in a catchment, they could potentially contribute to flood synchronisation issues. However, some studies highlight that leaky barriers only delay the flood peak rather than reduce it at a chosen location downstream. Most of the literature is based on modelling studies rather than empirically observed studies. This is owing to the short data records from leaky barrier experiments and the lack of study sites. The reduction can sometimes seem quite large, though the impact seems to be greatest at the local (reach) scale.

In addition, the uncertainties and assumptions in the modelling technique could play an important role (larger uncertainty in the output) as there is no standard modelling approach for representing leaky barriers documented in the literature. All methods adopt bespoke approaches applied to existing models and many of the studies note the uncertainties with their approaches. A general theme of the literature is scale. Most studies are performed at the local scale rather than catchment. This is not only an issue to do with data, but many studies also note the difficulties of installing leaky barriers in large rivers and keeping them in place. Generally, the literature has also highlighted that leaky barriers perform better in low magnitude (low return period) flood events rather than in extreme events.

A few commissioned reports agree that there are some remaining knowledge gaps surrounding engineered leaky barriers (SEPA and Forestry Commission Scotland 2012, Environment Agency 2014a) – see Section 2.4.8. For example, at an Environment Agency funded workshop in 2013, participants highlighted the need to carry out wider trials in a range of different catchment typologies testing different designs of leaky barrier (for example, from the more natural-looking measures to harder engineered approaches) so as to generate wider scientific knowledge and produce simple tools (Environment Agency, 2014a).

A few more studies have since been published and are addressed in this report, but many of the issues and knowledge gap remain highly relevant. Although there are a huge number of sites where leaky barriers have been installed (see Environment Agency and JBA mapping tools), nearly all of these case study sites are unmonitored. Those case study sites that are

being monitored on a larger scale (for example, Eddleston Water) (Figure 2.11) are yet to produce scientific evidence owing to short dataset lengths; promising results will hopefully be seen in the future.



Figure 2.11 Constructing a flow restrictor in the Eddleston catchment (© Tweed Forum)

The scientific literature surrounding leaky barriers (including naturally occurring wood accumulations and beaver dams, and engineered leaky barriers) suggests they can help to:

- increase hydraulic roughness (Gippel 1995, Environment Agency 1999, Abbe and Montgomery 2003, Curran and Wohl 2003, Montgomery et al. 2003, Manners and Doyle 2008, Kitts 2010, Dixon 2013, Valverde 2013, Ruiz-Villanueva et al. 2016)
- reduce flow velocities (Gippel 1995, Shields and Gippel 1995, Abbe and Montgomery 2003, Thomas and Nisbet 2012)
- increase the travel time of the flood wave (Gregory et al. 1985, Thomas and Nisbet 2012, Dixon 2013)
- create temporary storage (Shields and Gippel 1995)
- increase floodplain connectivity (Environment Agency 1999, Curran and Wohl 2003, Kitts 2010, Sear et al. 2010)
- attenuate flood flows (Forest Research 2008, Odoni and Lane 2010)

This list is based on both modelled and empirical studies.

2.4.3 Metrics of cost–benefit and multiple benefits

Naturally occurring large wood in rivers is a significant component of the form and function of river systems (Chin et al. 2014). In general, wood in rivers has many benefits. Wood in rivers provides the following benefits.

- It can help to collect sediments, sustaining gravel beds that are important habitats for fish.
- It creates areas of sediment scour and deposition, driving sediment sorting and spatial heterogeneity, landform building, and physical habitat complexity and turnover (Bilby 1984, Cherry and Beschta 1989, Abbe and Montgomery 1996, Addy and Wilkinson 2016).

- It promotes pool, riffle and bar formation (Gregory et al. 1994, Montgomery et al. 1995, Gurnell and Sweet 1998, Buffington et al. 2002, Bennett et al. 2008).
- It offers habitat for fish throughout their life cycle (Harmon et al. 1986, Roni and Quinn 2001, Kail et al. 2007, Mott 2010, Senter and Pasternack 2011, Dodd et al. 2016).
- It provides nutrients for aquatic organisms (Anderson et al. 1978, Nisbet et al. 2011b, Krause et al. 2014).
- It supports macroinvertebrate life cycles (Mott 2006, Cashman et al. 2016).

Generally, the same can be said for engineered leaky barriers in channels as these are designed to mimic natural processes. However, some dis-benefits may occur if a leaky barrier is designed incorrectly, for example, they are fixed in an incorrect configuration they may impact on migratory fish (see Dodd et al. 2016).

This section briefly reviews multiple benefits (and dis-benefits) behind both beaver dams, naturally occurring and engineered leaky barriers (focusing on key pieces of evidence from the list above).

Both naturally occurring and engineered leaky barriers may have positive and negative benefits for fish (Langford et al. 2012). In general, however, the literature suggests more positives (assuming engineered leaky barriers are designed correctly) (see systematic review by Stewart et al. 2006). In the River Blackwater in Essex, for example, it was found that fish populations increased around areas where leaky barriers were present. From 2007 to 2008, the total biomass of fish caught more than doubled with the number of roach in particular increasing (Environment Agency, undated a). However, further study is required in order to assess populations over following years.

A report by the Centre for Expertise in Water (CREW) sets out a series of guidelines for wood placement to ensure they do not impact negatively on fish (Dodd et al. 2016). The guidelines stressed:

- the need for robust management of each wooden structure placed in a stream (for example, checking them after a flood)
- the need for monitoring programme to be put in place to ensure fish passage is not affected by changes to structural integrity over time
- the potential for wooden structures to affect all fish species, with salmon and trout highlighted as species of most concern

The report defines wood placement in rivers according to 3 categories:

- placements that span the channel and are not in contact with water for the majority of the time (high flows)
- placements that span the river and are in contact with water the majority of the time
- placements that do not span the river (for example, on a channel bar)

It also points out that, if wood placement measures are placed too close to each other in series, this may have a negative impact on fish movement.

Focusing on aquatic habitats, wood placement in a fourth order stream in the New Forest increased habitat diversity by 46% (Kitts 2010). Flores et al. (2011) showed how wood placed in 4 streams in the Basque country in Spain increased the storage of organic matter 2–70 times. The Scottish Beaver Trial indicated that the number of plant species recoded in

lochs generally increased, especially those occupied by beavers (although the report suggested more data are needed). Beavers generally did not have an effect on plant species richness (Willby et al. 2014). However, the report indicated a significant negative effect of beavers on plant cover at the highest level of beaver occupancy. The report did not investigate the impact on flooding. Whitfield et al. (2015) suggested that increasing beaver populations could increase emissions of methane (a greenhouse gas) as beavers mediate aquatic emission of methane to the atmosphere by means of water impounded behind their dams. This, in combination with predicted increases in surface water temperature, and likely effects on the rates of methanogenesis, suggest that the contribution of beaver activity to global methane emissions may continue to grow (van Hulzen et al. 1999, Whitfield et al. 2015).

In relation to sediment management and geomorphic response, in-stream woody material denerally leads to reduced sediment transport (Jeffries et al. 2003, Dixon 2013), Marston (1982) found that up to 123% of the mean annual sediment discharge was stored behind log steps in the Oregon Coast Range. Similar results can be seen for beaver dams. Beaver ponds in Poland have experienced sedimentation rates of 14cm per year (Giriat et al. 2016). Engineered log jams in the Bowmont catchment in the Scottish Borders have been installed to trap coarse sediment on channel bars to mitigate coarse sediment problems. During one flood event, 16 structures induced acomorphic responses; however only 4 of the 33 structures induced significant deposition (> +0.3m), highlighting the importance of wood structure design and placement considerations (Addy and Wilkinson 2016). By increasing the size of engineered log jams and through careful placement that considers river channel scale, these structures can be designed to block more than 10% of the channel crosssection and increase the likelihood of significant hydraulic and geomorphic effects (Addy and Wilkinson 2016). Jeffries et al. (2003) found that the amounts (0-28kg m⁻²) and patterns of sediment deposition on floodplains with woody debris were both greater and more variable than had been observed on non-forested floodplains (case study: Highland Water, southern England). Naturally occurring wood in catchments of the New Forest created local points of flow avulsion and enhanced overbank sedimentation (Sear et al. 2010). Giriat et al. (2016) also found that 1,710m³ of sediment was deposited behind studied beaver dams between 2004 and 2011, with an average sediment thickness of 25.1 cm (De Visscher et al. 2014). This illustrates a higher sediment retention behind beaver dams than is typically observed behind wood jams. Such effective sediment retention drives the creation of beaver meadows and, more broadly, floodplains (Polvi and Wohl 2012) and provides the rationale for the use of beavers to restore incised rivers in some parts of the USA (Pollock et al. 2014).

A report from the Pickering catchment (Nisbet et al. 2011a) investigated ecosystems services created by leaky barriers and put a value on them, habitat and flood reduction score highly on this matrix (Figure 2.12). While agricultural profitability was found to decrease, aspects such as habitat and climate value increased massively.

Table 1: Indicative annual ecosystem service values based on central estimates for the woodland creation (85 ba) and LWD dam (150 dams) measures			
woodnand creation (o	Minimum (£/yr)	Maximum (£/yr)	Mean (£/vr)
Habitat creation	£0	£138,514	£121,524
Flood regulation	£4,200	£6,000	£5,964
Climate regulation	-£18,241	£317,943	£107,035
Erosion Regulation	£0	£221	£205
Education and knowledge	£10	£60	£14
Community development	£549	£549	£549
Agricultural production	-£32,056	-£3,771	-£31,604
Total	-£42,653	£431,180	£203,687

Table 2: Indicative ecosyste	m service present	values (£k at	2011	prices)	for	the
woodland creation (85 ha) and	LWD dam (150 d	ams) measures				

	Low (£k)	Central (£k)	High (£k)
Habitat creation	£1,630	£2,773	£4,459
Flood regulation	£88	£175	£292
Climate regulation	£923	£2,800	£5,464
Erosion Regulation	£0	£5	£10
Education and	£0	£1	£6
knowledge			
Community	£0	£16	£62
development			
Agricultural production	-£1,113	-£911	-£306
Forestry Costs	-£710	-£539	-£369
Net Present Value	£819	£4,321	£9,618

Figure 2.12 Tables depicting the ecosystem service values for woodland creation

Notes: The tables also consider woodland planting. Source: Nisbet et al. (2011a)

2.4.4 Effectiveness/performance

The previous sections have highlighted how all forms of leaky barriers can slow down the progression of a flood wave and reduce the peak magnitude. However, most studies have been performed at the local scale. The effectiveness and performance at the larger scale is unknown or subject to uncertainty. Notwithstanding this, a number of studies have pointed out that leaky barriers are appropriate only to smaller channels (stream orders of around 1 and 2) unless the channels are engineered. Utilising knowledge from across Europe, the European Commission's NWRM initiative has suggested that leaky barriers can be placed at scales of 0.1–1km² and that it is possible (with care) to install barriers at scales up to 1,000km².

Thomas and Nisbet (2012) found through a modelling exercise (using the 1 in 100 year design flood) that 5 LWD dams in a small Welsh tributary would reconnect the floodplain. They also found that the LWD dams reduced flow velocities by 2.1m³s⁻¹, thus delaying the flood peak by 15 minutes over a 0.5km reach. However, the effectiveness of LWD dams on larger scale catchments is still to be examined. In the Belford catchment in England, Nicholson et al. (2012) found that woody debris effectively forced high flows out onto the floodplain during times of high flow. The 8 woody debris dams were constructed from locally felled sycamore trees (Wilkinson et al. 2010a). Thomas and Nisbet (2012) found that wood placement measures slow the progression of flood waves, but they did not reduce peak magnitude. It was thought this was caused by water moving onto the floodplain and flowing back into the stream below the wood placement measures.

This trend of slowing the progression of floods has been noted elsewhere. Kitts (2010) showed through modelling that large wood accumulations in small-scale (~12km²) wooded catchments can alter the timing of a small magnitude flood peak by up to 33% New Forest case study). The Robinwood study also found similar results; leaky barriers can slow the flow but also attenuate floods. The study utilised a 1D hydraulic modelling approach and found, during a 1 in 100 year flood, that leaky barriers delayed the flood peak by a few minutes in a research catchment in Wales (Forest Research 2008). However, if the flood becomes too large and the features become submerged then the effects become less pronounced. The study highlighted how leaky barriers increased the connectivity of the stream to the floodplain thereby increasing the attenuation effect. Finally, the study notes that placing leaky barriers at the channel sources may be more effective as the constrictive effect is larger in smaller streams. This agrees with the finding of Quinn et al. (2013), who suggested that leaky barriers are best suited to headwater channels. However, it should be noted headwater streams usually have smaller in-channel storage and floodplains are smaller. The installation of leaky barriers in a river channel (as outlined in Dodd et al. 2016) usually has an immediate effect on the attenuation of flood flows compared with riparian planting, which takes time to establish. Therefore the combination of these 2 approaches can offer temporal NFM benefits.

2.4.5 Key case studies

This section summarises some key case study examples. For further details please refer to the case study documents in the <u>Evidence Directory</u>.

The NWRM website (<u>http://nwrm.eu</u>) highlights coarse woody debris as a WWNP measure. In the detailed⁹ summary of this measure, it presents only 2 case studies: Pickering in north Yorkshire and Belford in Northumberland. Both are in the UK and there are currently no other detailed European case studies. The website does highlight other 'light', less detailed

⁹ By detailed, the website refers to monitored studies.

case studies such as the Landscape Revitalisation Programme in Slovakia, which featured the installation of many leaky barriers (Kravčik et al. 2012).

As noted previously, there are a large number of unmonitored case studies. Also, there are a very small number of case studies that have solely installed leaky barriers for FCERM purposes. Instead, leaky barriers have been installed alongside an array of other measures. A summary of a selected few case studies that have installed leaky barriers primarily for FCERM purposes are presented below. Those case studies marked as conducting research are those where the study has been monitored in some way post construction and modelled.

- **Pickering, Yorkshire** [research]. Over 129 large timber debris dams were constructed in 66km² of catchment area. The water holding capacity of these dams ranges from 0.1m³ to 110m³ (depending on the size of the dam, together with the gradient and capacity of the stream channel upstream). The dams provide a total of 1,300m³ storage (based on a survey of 100 dams). The dams do not restrict fish passage, as low flows can pass beneath them (gap between bed and dams is typically 300mm). Each structure cost around £550–600 to build (based on a three-man team taking one day to construct) (Cronin 2016). The project found that 2 years is too short a timescale to demonstrate success, a reason why the project has fallen short in achieving some of the success criteria to date (Nisbet et al. 2011a). While 2 years may be sufficient for planning and case study work, a minimum of 3 years is recommended for implementing such a project.
- **Black Water, New Forest** [research]. Woody debris dams were installed over a 10km stretch. This work is summarised in previous sections.
- Stroud Rural Sustainable Drainage (RSuDS) project. Some 110 large leaky dams have been constructed along with 30 smaller, coarse leaky structures, which include in-channel deflectors and gully structures. This has been done over a catchment area of 235km². The project has limited data as many of the features are new.
- **Belford catchment**, **Northumberland** [research]. A total of 20 leaky barriers have been installed in a 5km² catchment, alongside an array of other measures.
- Eddleston Water Project, Scottish Borders [research]. A total of 79 leaky barriers have been installed over 69km² of channel (mainly headwater channels). The catchment area is 60km². No major scientific outputs are yet available covering the performance of the leaky barriers.
- **Bowmont catchment**, **Scottish Borders** [research]. Some 45 engineered log jams and 15 flow restrictors are located in the 80km² catchment. Flow restrictors are located in upland gulley catchments (less than 1km²) and engineered log jams are installed at varying scales (Figure 2.13). Addy and Wilkinson (2016) have shown that the scale at which these features are installed is important.
- **Restoration of River Lyvennet in Cumbria**. Strategy involves installing large wood material in the river to diversify the habitat and slow the flow.
- Holnicote Flood Management Demonstration Project. Leaky barriers have been installed to help deflect flows into floodplain storage features, and a large number of leaky barriers have formed naturally in Horner Woods following the cessation by the National Trust in 2007 of the practice that had previously removed all woody material from the Horner Water channel.



Figure 2.13 Engineered log jam with planted trees in the middle of the structure on a gravel bar in the Bowmont catchment, Scottish Borders (© Steve Addy)

2.4.6 Funding

The cost of installing a leaky barrier is highly variable, as it is related to factors such as size of river, accessibility and level of engineering required. SEPA's 'Natural Flood Maanagement Handbook' gives a cost range of £100 to £1,000 to install a typical wood placement feature, including felling and installation costs (SEPA 2016). This is a similar cost to that quoted by Quinn et al. (2013). However, this cost is dependent on the amount of engineering required to install the feature (including pre-works assessments). If a site is harder to access then the cost could be much larger. However, more traditional engineering approaches can be used. For example, the Haltwhistle Burn wood placement matrix was installed in a difficult to access forest; as it was difficult to get machinery into the forest, horses were used instead (Henderson 2015). This also reduced the impact of soil compaction. The Stroud RSuDS project was able to construct around 110 leaky barriers costing around £1,500 each. The Bowmont bar apex engineered log jams cost around £230 each to construct (Addy and Wilkinson 2016).

Very little of the literature has covered funding arrangements. The SEPA NFM handbook does cover funding arrangements (SEPA 2016), highlighting agri-environmental schemes and the Water Environment Fund¹⁰ as important sources for funding. A number of studies have been funded via the European Union though LIFE and Interreg grants.

The Environment Agency has developed guidelines on the whole life costing of a FCERM project (Environment Agency 2015a), accompanied by a suite of notes covering all aspects of FCERM.¹¹ However, the report acknowledges there is limited information on whole life costing for a WWNP based project (specifically leaky barriers) and hopefully further information may come from information presented in the case studies above.

A report commissioned by the Scottish Government assesses the mechanisms for compensating land managers (RPA et al. 2015). The aim was to investigate the options for compensating land managers who implement NFM measures on their land and the report identifies a range of mechanisms by which public bodies can compensate land managers, including land purchase/sale, lease back and wayleaves. The report also presents a 5-step plan for developing a payment rate for compensation.

¹⁰ <u>https://www.sepa.org.uk/environment/water/water-environment-fund</u>

¹¹ Project SC080039. See http://evidence.environment-agency.gov.uk/fcerm/en/Default/FCRM.aspx

2.4.7 Design, management and maintenance

This section refers to the management of engineered leaky barriers. As detailed above, naturally occurring wood in rivers can have a number of benefits and in most cases should be left in the channel (unless it may create issues for an asset). Please refer to reviews by Environment Agency (1999) and Section 3.4 on riparian planting for further details. At the time of writing, the Eurasian beaver is only present at key experimental sites in England and Scotland. Please contact the local regulatory body about the status of the beaver in the UK.

Engineered leaky barriers – and to an extent naturally occurring wood in rivers – can be complex to manage. Incorrectly placed wood can result in increased flows around and underneath the structure, increasing bank and bed erosion locally. It can also have conflicts with human activities such as canoeing, but more importantly could have an impact on migratory fish (see Dodd et al. 2016). However, well-placed wood can substantially boost the ecological functioning and biodiversity of watercourses (Krause et al. 2014).

There are concerns that leaky barriers should be fixed to the banks and/or bed to ensure the wood does not become mobile in a flood. Mobility is a serious issue to many managers (see, for example, Chin et al. 2014). Dixon and Sear (2014) highlighted the mobility of wood in the New Forest, with over 75% of the pieces of wood moving in the study period. However, very large pieces of wood (greater than 2.5 times the channel width) should be considered as functionally immobile (Dixon and Sear 2014). A study in Sussex suggests that wood should be at least 1.5x the channel width (The Sussex Flow Initiative 2016), though this rule may be impractical for large rivers. However, many items in the literature identify log movement issues if wood is placed in large or energetic river systems (Ruiz-Villanueva et al. 2013) and highlight that further research is needed to understand the proximity of leaky barriers to a vulnerable downstream flood risk receptor or asset (for example, a bridge or culvert). Therefore not all streams lined with woodlands are applicable for in-channel wood measures.

Scouring may occur shortly after construction resulting in the settlement of the trunks (Quinn et al. 2013). Quinn et al. (2013) noted that these features should be inspected after a flood in case blockages and unexpected scouring have occurred. It is possible that scour and deposition cycles will find a more natural balance in time. If scouring becomes excessive remediation will be required. Scouring rates are dependent on the structure design and the stability of the banks and bed so will vary from site to site. Therefore consultation with a fluvial geomorphologist is required in the planning phase. Although wood will decay naturally, the rate of decay depends on the type of wood, the size of wood, how often the wood is wet, the temperature and the setting (Lofroth 1998). Dixon (2013) noted that information on decay rates for wood in rivers is sparse.

There are a few simple guidance documents already in place for leaky barriers, though they do not go into full technical designs. The guidance document from the Sussex Flow Initiative (2016) identifies the following as the best places to locate wood placements in streams:

- disconnected floodplains
- headwater streams
- streams lined with woodlands
- drainage ditches
- degraded uniform channels
- areas adjacent to flood storage areas

The guidance also notes the need to secure wood. It also highlights that:

- barriers should not be installed close to an urban area
- wood should be untreated
- local materials should be used where possible

The placement of any structure in a watercourse (see example in Figure 2.14) has the potential for both beneficial and adverse effects on habitats and flows (SEPA 2016). With this in mind, SEPA NFM handbook sets out a number of criteria that should be followed when installing leaky barriers (SEPA 2016). Before installation there is a need to:

- make a detailed hydromorphological assessment of the proposed river reaches, taking into account sediment dynamics and effects on flood risk (this is effectively a fluvial audit for the catchment or subcatchment in question)
- to carry out hydrological and ecological monitoring before and after construction (Dodd et al. 2016, SEPA 2016)
- consider the structure design and stability in relation to project aims, river dynamics and material used (Brooks 2006)
- conduct a habitat survey (both in-channel and on the riparian zone) to ensure no habitat could suffer damage (including during the construction phase)
- check for preservation orders, nesting birds or bats if trees are to be felled
- consider adjacent habitat and the suitability of change as a result of a leaky barrier location
- model various locations of one or multiple barriers and types
- monitor and survey fish passage

Nisbet et al. (2011a) noted that it may be more difficult to implement leaky barriers at a larger catchment scale, where the river channels are larger and there is a greater risk of wood washing away and blocking bridges downstream. Indeed, naturally formed, channelblocking wood jams are mainly confined to smaller channels, where the channel width is less than the typical tree height, although very large valley jams occasionally occur in larger channels (Abbe and Montgomery 2003). Thus, in larger channels, engineered wood structures are usually secured in place to prevent them moving, making them very different from natural wood structures. Installing a trash screen at the downstream end of a series of leaky dams could be a solution to the wood mobility problem (Wilkinson et al. 2010b).



Figure 2.14 Wood placement measure, Belford Burn, Northumberland

Notes: Structure attenuates high flows while not restricting low flows. Source: Mark Wilkinson

Guidance from the Environment Agency (Figure 2.15) is based on similar principles to those outlined in SEPA (2016) and in the list above.



Figure 2.15 Flow chart to help decide whether to install a leaky dam

Source: Environment Agency internal operational instruction (2011)

2.4.8 Research gaps

There is a vast amount of knowledge surrounding leaky barriers. However, the literature has also suggested there are a number of research gaps that still need to be addressed to fully understand how leaky barriers alter hydrological processes during floods. Also, there are other gaps, for example, around management issues. Known gaps are summarised below.

Understanding of effectiveness of leaky barriers

- Despite an enormous scientific literature on the presence, dynamics, hydraulic, geomorphological and ecological roles of wood in rivers (for reviews see Gurnell et al. 1995, Gregory et al. 2003, Gurnell 2013, Wohl et al. 2016), evidence concerning the role of wood barriers in relation to flood risk is limited (SEPA and Forestry Commission Scotland 2012, Environment Agency 2014a). Nevertheless, this literature supports current guidance on leaving or placing woody debris in river channels (Environment Agency undated a).
- A general theme of the literature is scale. Most studies are performed at the local scale rather than catchment scale. A gap remains about the effectiveness of leaky barriers in mitigating flood peaks at the catchment scale.
- Most studies consider riparian planting with leaky barriers together. Therefore there are a limited number of studies that solely consider the impact of leaky barriers. There is a need to understand both the role of leaky barriers in isolation and to what degree they assist other measures such as floodplain restoration.

- There is not a specially designed tool for modelling leaky barriers. Most tools take a bespoke approach to existing models using assumptions or features that may not be applicable to leaky barriers. For example, how do we correctly use parameters such as Manning's n?
- Linked to the above, participants at an Environment Agency funded workshop in 2013 highlighted the need to undertake wider trials in a range of different catchment typologies testing different designs of leaky barrier (for example, from the more natural looking measures to harder engineered approaches) so as to generate wider scientific knowledge and produce simple tools (Environment Agency 2014a).
- There are a huge number of sites where leaky barriers have been installed (see Environment Agency and JBA mapping tools); however, nearly all these case study sites are unmonitored. The case study sites that are monitored on a larger scale are yet to produce scientific evidence owing to short dataset lengths (for example, Eddleston Water), although promising results will hopefully be seen in the future.
- More research is needed to understand how beavers could be used to mitigate flood risk in the UK, however, local trials are producing interesting findings (see, for example, Puttock et al. 2017).

Management of leaky barriers

- There is a gap around the understanding of the liability of engineered leaky barriers. Should a debris dam be dislodged and increase downstream flood risk, who is liable?
- Further information on whole life costs and engineering performance is required.
- There are knowledge gaps related to the management of natural large wood and river corridors including: an analysis of the consequences of enormous historical reductions in large wood load in rivers through the forested portions of the temperate zone; and how to effectively reintroduce and manage existing wood in rivers, which includes enhancing public understanding (including relevant stakeholders) of the importance of wood in rivers (Wohl et al. 2016).
- Questions still remain around the checking and maintenance of these structures after a flood. For example, how should a proper check be undertaken? Who should undertake the check? What is the checker looking for? How does the checker know when something is wrong and requires action? These questions highlight the need for guidance around the maintenance of leaky barriers after construction.
- A report from the Beaver Salmonid Working Group (2015) suggested that there are significant gaps in our knowledge of beaver–salmonid interactions. The report also suggested that, if beavers were introduced, a detailed management plan is required liaising with all interested stakeholders.
- There is a need to further engage with all stakeholders with regard to the use of beavers for FCERM (for example, asset managers, farmers and fishermen).

2.5 Offline storage areas

Before reading this review of offline storage areas, we recommend first reading Section 2.2 on floodplain restoration.

Offline storage areas are floodplain areas that have been adapted to retain and attenuate floodwater in a managed way. They usually require the construction of a containment bund, which elevates the amount of water that can be stored on a floodplain and will usually require an inlet, an outlet and potentially a spillway mechanism.

There are many different terms used internationally for describing offline storage areas, however, the most important difference between these is the size and engineering involved in the design. For example, the terms 'washlands' (Morris et al. 2004b) and 'polders' (European Commission 2015b) have been used in the literature, though it should be noted that these features are large scale and can store more than 10,000m³ (sometimes significantly larger orders of magnitude). This review refers mainly to washlands rather than polders owing to their applicability in the UK landscape. The term 'run-off attenuation feature' (RAF) was used by Wilkinson et al. (2010a) and Quinn et al. (2013) to refer to small-scale offline storage areas in subcatchments smaller than 10km² (in this case the Belford catchment in Northumberland). However, Metcalfe et al. (2016) applied the term 'run-off attenuation features' in the context of a larger catchment in north Yorkshire (although only through modelled scenarios, no features have been installed).

Washlands are defined as managed floodplains that are allowed to flood. These floodplains may be deliberately flooded by a control structure for reasons such as flood attenuation or the creation of new habitat (Hardiman et al. 2009).

A polder is a common offline flood storage measure used in the Low Countries of Europe (Belgium, Luxembourg and the Netherlands). European Commission (2015a) defines the measure as:

'a low-lying tract of land enclosed by embankments (barriers) known as dikes that forms an artificial hydrological entity; it has no connection with outside water other than through manually operated devices. Its re-naturalisation consists in enhancing polders with sub-natural characteristics, allowing better water storage in watercourses inside the polder, as well as increased biodiversity'.

Polders are usually employed within large0scale catchments (for example, the Rhine) at scales of 100–1,000km² and have more engineered hydraulic structures associated with them. Washlands can vary in size and the level of engineering required (see Figure 2.16). The important distinction between a washland and a polder is that some washlands can operate naturally and do not require engineering such as manually operated devices.

Generally, the common theme between all these features (washlands, offline ponds, polders) is that they are man-made landscape features whose main purpose is to hold water more efficiently and in a controlled way on a floodplain. However, the scale of both the measure (in terms of water holding volume) and catchment area where they are placed can vary substantially. This review looks specifically at the literature behind small-scale offline storage areas (for example, RAFs and small washlands) through to larger scale washlands. However, it tries to address the literature behind those measures which function under the remit of WWNP, that is, the feature in question is not heavily engineered and therefore falls under the traditional storage area literature (that is, traditionally engineered flood defences).

Figure 2.16 presents a hydraulic matrix for classifying washlands by the degree of hydraulic control. Or to put it in the context of this review, the degree of engineering. Therefore, washland types 1, 2, 4 and 5 are covered in this review, but not types 3 and 6–9 (Figure 2.16). The bottom right of the matrix falls into the remit of traditional engineering;

while an aspect of 'soft' engineering is needed for washland types 1, 2, 4 and 5, this is becoming more complex towards type 5.

This review focuses on offline storage areas which function more naturally (within the WWNP remit). For a detailed review of traditionally engineered flood storage areas (including online) and washlands that adhere to traditional flood storage principles, please see Environment Agency (2016c).

		Inflow		
		Uncontrolled inflow	Fixed controlled inflow	Variable controlled
				inflow
	Uncontrolled	1	2	3
	gravity return	As river stage rises, water	Water flows into the	Water is let into the
		flows onto the washland	washland once a flood	washland via a sluice gate
		and returns to the channel	bank is overtopped, and	at the discretion of the
		when the stage falls. This	returns to the channel in	flood manager, and
		situation is akin to a	the same vicinity via a	returns to the channel via
		natural flood plain and is	flapped outfall when the	a flapped outfall when the
		the best example of on-	stage falls	stage falls.
		line storage. Examples		
		include the Long Eau and		
		Steenwaard		
		(Netherlands).		
	Fixed controlled	4	5	6
	gravity return	This situation is unlikely	Water flows into the	Water is let into the
		to occur as if water flow	washland once a flood	washland via a sluice gate
		into the washland is	bank or spillway is	at the discretion of the
		unimpeded return flow	overtopped. Water	flood manager, and
		should also be	returns to the channel	returns to the channel via
M		unimpeded.	back over the	a flapped outfall some
ťĮ			embankment /spillway or	distance downstream
On			via a flapped outfall some	where there is sufficient
-			distance downstream	head difference for
			where there is sufficient	gravity flow. Examples
			head difference for	include the Alterheim
			gravity flow. Examples	Polders (Germany).
	37 11	-	include Coombe Hill.	0
	variable	·/	NV O i i d	9 W. O
	controlled	I his situation is unlikely	water flows into the	water flows into the
	return	to occur as if water flow	washland once a flood	washland when a control
	(sluices/pumps)	into the washiand is	bank is overtopped, and	on the river is closed (at
		unimpeded return flow	is then pumped back into	the discretion of the flood
		should also be	include Declain cham	manager), and returns to
		unimpeded. It could be	Maash	the channel once the
		could enter via a flanned	IVIAI SII.	Examples include
		could enter via a happed		Harbertonford and the
		flow and is then pure ad		Loigh Porrier
		hack into the river but		Leigh Darrier.
		this example was not		
		found		
		iouila.		

Figure 2.16 Hydraulic matrix classifying washlands by degree of hydraulic control

Source: Morris et al. (2004c)

2.5.1 Understanding the science

The storage and attenuation of floodwater is the main mechanism by which offline storage areas reduce downstream flood risk (as discussed in Section 2.1). Therefore, there are 2 main challenges for managers. The first is to be able to control the inflow and outflow of water to ensure offline storage areas have sufficient flood storage capacity when it is most needed. Second, managers need to recognise the importance of ensuring sufficient storage is available by selecting a site that can retain sufficient water and recognising that, if the

offline storage area is reduced (that is, it is already storing water), then the flood defence facility is decreased (Morris et al. 2004c).

Quinn et al. (2013) utilised a Pond Network Model to determine the effect of increasing the number of offline storage ponds located along a river reach on attenuation in the Belford catchment in Northumberland (Figure 2.17). Each pond represented in the model was given a 550m³ storage capacity and the upstream catchment area was around 4km². Each line in Figure 2.18 represents the addition of a new pond until 35 ponds with nearly 20,000m³ capacity are added. The peak flow reduction is estimated to be between 15% and 30% for both observed storm events and Flood Estimation Handbook (FEH)¹² design storm events. It has been demonstrated using empirical evidence in a modelling framework that reductions in peak flows can be achieved through storage and attenuation of water by using a cascade of these features in the drainage network (Quinn et al. 2013).



Figure 2.17 An example of an offline pond in the Belford catchment

Source: Quinn et al. (2013)

When the model was applied to the Belford catchment, Quinn et al. (2013) found that the discharge peak of the largest flood event (March 2010) was only significantly reduced (by ~5%) until ~10,000m³ of storage was added to the network. They attribute this finding to the fact that most of the ponds filled before the arrival of the main flood peak. This emphasises the importance of understanding the critical number of offline ponds needed to create significant peak flow reduction (by ~5%) in other catchments of a similar size.





Notes: Pond Network Model made use of empirical discharge data.

¹² <u>https://www.ceh.ac.uk/services/flood-estimation-handbook</u>

However, a major assumption of the study was that it is possible to install 35 ponds in a network within the riparian corridor in such a small catchment. It is likely this will be challenging to achieve in practice owing to insufficient floodplain space and the requirement for sufficient farmer uptake of the approach. Depending on the land use and lands-soil management in the upstream catchment, ponds can also fill up quickly with sediment. This can benefit the water quality of downstream water bodies. However, this sediment needs to be periodically removed to maintain the overall storage capacity of the pond (Barber and Quinn 2012, Wilkinson et al. 2014).

Five floodplain offline storage ponds were created in the Aller catchment on the Natural Trust's Holnicote Estate in west Somerset by constructing earth bunds with piped outlets. These ponds can hold up to about 25,000m³ of additional floodwater on the floodplain (above the natural flood storage volume that is available). A 1D–2D hydraulic model was used to assess the attenuation effect of these ponds. The model was driven by data collected during storm events as well as FEH derived design event data. Using data from the 23–24 December 2013 flood event, there was a decrease in peak flow from 14.4m³s⁻¹ to 13.1m³s⁻¹ – a 10% reduction in the flood peak for a catchment area of ~15km² (National Trust 2015). The reduction may be larger for floods with a smaller return period (for example, a 1 in 5 year event could see a 25% reduction).

2.5.2 Confidence in the science

'In England when we see flooded fields we see it as a sign of failure. In Holland it is a feature of a working water management system' (Hickman et al. 2001).

This statement goes to show that offline storage areas, whether natural or humanconstructed features, are viewed internationally as a FCERM approach.

There are fewer studies for smaller scale storage areas, but these show that storage can lead to positive FCERM outcomes (see, for example, Quinn et al. 2013). Out of all the catchment-based measures presented in this review, offline storage areas are probably the most engineered (especially as size increases) and can be costly to install (and compensate for). An individual large-scale storage measure can make a significant impact on FCERM (see, for example, Metcalfe et al. 2016). However, the Making Space for Water report identifies the importance of land management in washlands that integrates habitat with flood management report (Defra 2004a). Morris et al. (2008) concluded that it is feasible to create zones in the floodplain that can store water and potentially meet all or some of the objectives of flood and water resources management and environmental enhancement, and support the rural farming community (though some form of compensation is usually required).

Quinn et al. (2013) stated that, although individual small-scale RAF storage measures contribute to flood attenuation (albeit small), their effectiveness for FCERM lies in understanding how they integrate into the hydrological response of the entire catchment area (as found in the ~5km² Belford catchment in Northumberland). However, Quinn et al. (2013) acknowledged the need for further work to upscale the findings (that is, how much storage is needed in larger catchments?).

Offline storage ponds should be designed to fill and empty at a set rate and the size of the outlet pipe is critical for controlling the flow. Ideally, a 550m³ storage pond within a 4km² catchment should empty within ~12 hours to provide storage for following events and, if not located on a buffer strip, to reduce damage to farm crops (Wilkinson et al. 2010a).

Clearly, larger catchments require a larger offline storage volume if flood risk is to be noticeably reduced. The approach traditionally adopted by engineers has been to create one large storage area whenever possible and manage this as a traditional flood storage area. Although the principles are similar, this level of engineering goes against the WWNP philosophy. There are some examples of large-scale (for example, around 100,000m³) storage washlands in the UK that have been created in large catchments (for example, >100km²) and which fall under the remit of traditional engineering (see Section 2.5.5. for a brief description of these). However, there remains a gap in the literature on the impact of using many smaller scale offline storage measures (such as RAFs and small washlands, such as those used at Holnicote) distributed across large catchment areas (for example, taking the work of Quinn et al. 2013 and Metcalfe et al. 2016, and applying these measures over larger scale catchments) and comparing how these function as opposed to one large storage area.

The concept of NFM has been discussed by Quinn (2016a), who suggested that the larger volumes of storage needed in larger catchments, such as the Eden, should be achieved by distributing many smaller measures across the landscape (coupled with traditional engineering). Following on from Wilkinson et al. (2013a), there is also a need for a fuller understanding of the Reservoirs Act in relation to offline storage features. For example, does a series of measures in sequence fall under the act?

2.5.3 Metrics of cost–benefit and multiple benefits

Offline storage areas, if managed appropriately, have great potential to support biodiversity through the open water and wetland habitats that they create (Morris et al. 2004c, eftec 2015).¹³ Sendzimir et al. (2014) pointed out that a network of ponds as opposed to one large reservoir provides a greater diversity of migration pathways for avian, amphibian, mammal and reptile species. Offline storage areas also have the ability to improve water quality by capturing sediments and cycling nutrients and pollutants (Morris et al. 2004c).

The English Nature report on washlands noted that the opportunity to integrate biodiversity needs into a washland depends on the ability to maintain wet conditions beyond the period of flood events. This ability depends on the land use and the farmer's needs (and ultimately in some cases, more compensation). Where a washland is used for arable cropping, the scope for habitat enhancement is low, but the scope is higher if a washland supports grassland or woodland land use. The report also stated that there are opportunities for synergies if a washland is overdesigned for flood storage, thereby providing extra capacity to support biodiversity (Morris et al. 2004c).

Literature that considers the value of washlands is quite limited and mainly relates to the value of wetlands (see Section 2.2). However, the RESPONSE project (Sendzimir et al. 2014) examined the cost-effectiveness and multiple benefits of on-farm NWRM as an alternative to traditional investments in dams and reservoirs. Their preliminary results show that 'green infrastructure', especially conservation tillage and (to a lesser extent) tree shelterbelts, are cost competitive with 'grey infrastructure' like dams and reservoirs for retaining water. In addition, they contribute to climate change mitigation, biodiversity and pollination improvements, and reduce land degradation.

Eftec (2015) calculated the natural capital account of the Beam Parkland washland in London (see case studies) (Figure 2.19), estimating that this washland provides £770,000 per year in community benefits and £591,000 per year in flood reduction benefits as well as significant (but uncosted) biodiversity benefits. The literature suggests that establishing a washland on a floodplain can manage flood risk and support the rural economy through the provision of a range of other ecosystem services (Morris et al. 2004b, Morris et al. 2004c, Everard and McInnes, 2013).

Typically, there is a trade-off between the needs of agriculture and biodiversity for offline storage areas (particularly large-scale washlands). To create richer and more diverse

¹³ The report by Morris and colleagues for English Nature focuses on washlands.

habitats, washlands should remain wetter for longer, but this may conflict with the agricultural needs of a washland (Morris et al. 2004c). A recommendation by Morris et al. (2004c) was to promote guidance and training for flood management and biodiversity objectives so they can be delivered simultaneously.



Figure 2.19 Beam Parkland natural capital account

Source: eftec (2015)

2.5.4 Effectiveness/performance

Quinn et al. (2013) and National Trust (2015) have shown that offline storage areas have positive FCERM benefits. The previous sections have highlighted the performance of these measures. Eftec (2015) described how a large-scale washland near Dagenham (Beam Parkland) has positive benefits for FCERM. The Beam washland can hold up to ~460,000m³ of floodwater and provides a standard of protection to downstream properties for up to a 1 in 25 year event (eftec 2015). The added pumping station within the washland provides an enhanced standard level of protection of up to 1 in 150 years.

In the Tarland catchment in Aberdeenshire, a combined 1D (for storage modelling) and 2D (for flow routing and flood inundation) hydraulic model was constructed to explore the effectiveness of offline storage area scenarios (Ghimire 2013, Ghimire et al. 2014). The catchment has a dense monitoring network but, owing to the short length of the time series data, the FEH methodology was used to drive the model. The results show that a single pond storing 27,000m³ attenuates a 1 in 2 year event (QMED) by ~9% (for a catchment area of 25km²) (Ghimire et al. 2014). However, the study utilised stakeholder feedback which supported the concept of many small storage areas rather than one large (27,000m³) storage area. Therefore several smaller ponds (ranging from 1,500m³ to 4,000m³) were incorporated into the modelling framework at opportunistic sites to give a total storage of 23,000m³). Results for this scenario show that the QMED event was attenuated by ~5% which is comparable with the large offline storage area scenario. Ghimire et al. (2014) found

that, if these smaller areas were made slightly larger (4,000–6,000m³); giving a total storage of 48,000m³, then the QMED event could be attenuated by ~12%. This study illustrates that a network of smaller features can offer, in some instances, improved flood peak attenuation compared with the creation of one large feature. The Tarland case study provides a valuable insight into the performance of measures at a catchment scale. However, the model was not informed by empirical data and its authors wish to further develop it with data from recently installed hydrometric stations. It should be noted that only one storage area has been created in the Tarland catchment. Owing to its size and position in the catchment, the feature is unlikely to attenuate large flood events (see Section 2.5.5).

Washlands offer an effective way to store water on floodplains in a controlled manner that can reduce downstream flooding (Morris et al. 2004c). However, there is little uptake of washlands or managed floodplains in the UK over large-scale areas. According to Hickmann et al. (2001), the main reason why planners do not consider washlands to be a viable option reflects a lack of imagination by operating authorities and consultants undertaking the appraisals. Since Hickmann et al. (2001) was published, a number of new washland and smaller offline storage area sites have become operational in the UK (see case study list). However, many of the arguments made by Hickmann et al. (2001) remain valid. For example, Guerrin (2015) found that institutional factors played a critical role in the failed delivery of a washland restoration project on the River Rhône, France.

2.5.5 Key case studies

Owing to the trade-offs in washland management, Morris et al. (2004c) compiled a list of case studies that included projects with mainly nature conservation or flood management goals as well as those integrating both motivations. Some of the more naturally functioning washland types are outlined below.

Mainly conservation washland

- Long Eau, Lincolnshire [unmonitored]. This site is flooded around 3–4 times a year. Water can remain on the site for a few days to months. The inflow and outflow are uncontrolled. This is an example of a naturally functioning washland.
- **Coombe Hill, Gloucestershire** [unmonitored]. The site has an area of ~650ha and is flooded by the River Severn. Flooding occurs annually over much of the site and water can remain on the washland for around 2 weeks. The land use is grassland. The site offers limited FCERM benefits.

Both these sites are located on clay soils with a relatively low hydraulic conductivity, which restrict any significant infiltration losses (Morris et al. 2004c).

Other offline storage area case studies

- **Belford, Northumberland** [monitored]. Five offline storage areas have been created in the catchment (6km²) ranging from 400m³ to 2000m³. The performance of these measures is described above. Modelling has shown this approach to be effective, especially when the density of storage areas increases (see Nicholson et al. 2012).
- Holnicote, Exmoor [monitored]. Offline flood storage bunds were created in the catchment to store water more efficiently on the floodplain. These ponds offer up to about 25,000m³ of additional storage on the Aller floodplain (National Trust 2015).

- Farming Floodplains for the Future, Staffordshire [unmonitored]. This project sought to create floodplain storage ponds as well as reconnecting the river to the floodplain. With a focus on reducing flood risk to the town of Stafford, the target storage volume required was 435,000m³. The project managed to deliver 2% of this target, creating 9,000m³ of storage on floodplains. For the town of Penkridge, a total of 13,000m³ storage was created upstream (3.2% of target). However, a major success of the project was the engagement with farmers and their interest in helping to reduce the flood stream flood risk. The project estimated the average cost per cubic metre of storage to be about £5 (Jones 2010).
- Metcalfe et al. (2016) has explored offline storage in the **Brompton catchment**, **north Yorkshire**. No measures are in place yet, but the authors have implemented scenarios into a modelling framework and are currently assessing the impact of these measures.
- **Tarland, Aberdeenshire** [monitored]. This offline storage area was constructed at the Mill of Gellan on the Tarland Burn. The feature has a capacity of 2,000m³ and is located at a scale of 54km². However, the feature demonstrates the concepts of offline flood storage and the feature alone will have little FCERM benefits for a large flood event (more features are needed in the catchment). A modelling scenario was developed for the catchment which shows the potential for offline flood storage to mitigate flood events in the catchment (see above) (Ghimire et al. 2014).

Table 2.1 reproduces a list of further case studies taken from a report complied for English Nature in 2001.

Name	Location	Brief description
River Calder	Wakefield, Yorkshire	Washlands are an integral part of the preferred flood defence option being progressed to construction.
Haddiscoe Island	Norfolk	Saline washlands considered as part of Broadland Flood Alleviation Strategy but not taken forward as preferred option.
River Witham	Lincolnshire	Agreed flood alleviation strategy included four washlands.
Ouse Washes	Cambridgeshire	Existing Ouse Washes suffering from summer flooding which affects the flood carrying capacity. Additional washlands being explored as an option to attenuate summer flood flows.
River Ancholme	Lincolnshire	Washlands considered as an option but was not the preferred option as provision of hard defences was considered more cost effective.
Melton Mowbray Flood Alleviation Scheme	Leicestershire	Washlands part of the agreed flood alleviation scheme to be implemented shortly.
Lincoln Flood Alleviation Scheme	Lincolnshire	Washlands to protect the city of Lincoln have been in operation for 17 years.
River Trent Beckingham Marshes	Gainsborough, Lincolnshire	Washlands as part of flood defences have been in operation since 1960.
River Nar	Norfolk	Washland option considered in long list of flood defence options but not taken to detailed assessment due to high cost and technical concerns.

 Table 2.1
 Summary of example UK washland projects

Source: Hickmann et al. (2001)

The Wakefield Flood Alleviation Scheme on the River Calder, which is classified as a traditional flood storage area scheme, implemented washlands as part of the FCERM measures to protect downstream homes and businesses (Environment Agency 2000). The washlands were based on the use of 5 upstream storage sites providing a 1 in 25 year level of protection (Hickmann et al. 2001). Washlands were also implemented on the River Witham and the tangible benefits of the preferred option amounted to £533 million (together with embankment strengthening, a 1 in 25 year standard protection level was achieved) (Bullen Consultants 1997). At Melton Mowbray, washlands provide tangible benefits amounting to £14 million (present value) with a 1 in 100 year standard level of protection (Environment Agency Midlands Region 2000); Hickmann et al. (2001) noted that at this site compensation to landowners amounted to £240,000. An additional £173,000 was also invested in environmental benefits (for example, wetland creation within the washland).

One further example is the Beam Park washland, Dagenham, London. Although an urban washland within an area designated in the Green Infrastructure Plan for Greater London, it should still be considered for this review as many of the FCERM principles are similar to non-urban storage areas. The washland has a capacity of ~460,000m³. It is drained by a spillway or, if flows exceed the spillway, pumps can be operated (eftec 2015).

2.5.6 Funding

There are a number of ways to manage and administer offline storage area creation and operation. For example, for larger scale areas, through land purchase, easements on flooding, management agreements supported by annual payments and leaseback partnership arrangements (Morris et al. 2008). Generally, the funding arrangements are handled by the Environment Agency planning teams. There is little literature assessing the precise funding arrangements for individual schemes as these can change year to year. There are a number of examples of the use of funding partnerships, whereby different funding sources, some of which may even have been individually justified on non-flood risk management grounds, are drawn together to deliver an integrated project or scheme which delivers multiple benefits.

Other potential funding opportunities include:

- Heritage Lottery Fund
- local authority capital grants
- Flood Defence Grant in Aid or Local Levy
- agri-environment schemes
- biodiversity offset schemes
- landfill tax credit schemes
- charitable funds

The Environment Agency has developed guidelines on the whole life costing of a FCERM project (Environment Agency 2015a), accompanied by a suite of notes covering all aspects of FCERM.¹⁴ However, the report acknowledges there is limited information on whole life costing for a WWNP based project (specifically small-scale offline storage areas) and hopefully further information may come from information presented in the case studies above.

¹⁴ Project SC080039. See <u>http://evidence.environment-agency.gov.uk/fcerm/en/Default/FCRM.aspx</u>

A report commissioned by the Scottish Government assesses the mechanisms for compensating land managers (RPA et al. 2015). The aim was to investigate the options for compensating land managers who implement NFM measures on their land and the report identifies a range of mechanisms by which public bodies can compensate land managers, including land purchase/sale, leaseback and wayleaves. The report also presents a 5-step plan for developing a payment rate for compensation.

2.5.7 Design, management and maintenance

An offline storage area/washland usually has an inflow, an outflow and a spillway.

The inflow mechanism can be (Morris et al. 2004c):

- uncontrolled (for example, natural spilling from a river which is not artificially controlled)
- fixed controlled (for example, a set sized embankment or inflow pipe which floods the storage area)
- variable controlled (for example, a sluice gate)

The outflow mechanism can also be classified in 3 ways (Morris et al. 2004c):

- an uncontrolled gravity return (for example, water can flow back into the river as the river level drops)
- fixed controlled gravity return flow (for example, as the river level drops, water can return to river via a flapped outfall)
- a variable controlled return outfall (for example, a sluice mechanism controls outfall rate)

Morris et al. (2004c) identified 3 main types of management funding arrangement:

- land purchase from existing owners
- purchase of flood easements
- use of management agreements (linked to an agri-environment scheme)

They also pointed out that purchased land could be leased back to the owner.

Figure 2.20 shows an offline storage pond on a floodplain buffer zone during and after a flood event.

According to the European Commission's NWRM initiative, maintenance costs are generally 0.5–1.5% of the initial implementation costs. However, most large storage areas referred to in this review are usually not located on farmland. They are designated areas that have usually had the land purchased for the purpose of flood storage. Maintenance of constructed storage areas are usually maintained by the Lead Local Flood Authority. The literature suggests smaller storage areas such as RAFs are located on farmland and generally work alongside farming practices.

Where an offline storage area is located on farmland, it is important to recognise that arable crops are more susceptible to damage from waterlogging than grasslands. Therefore, it may be preferable to work with grassland systems first. If the washland or offline storage ponds are on more favourable agricultural areas, there may be a requirement to manage soil water to ensure crops are able to grow on the site after the floodwaters have receded. Morris et al. (2004a) summarised the estimated flood damage costs on improved grassland based on the

duration and depth of flooding during winter months. Short duration flooding (1–2 weeks) has little impact on most grasslands costing about £15 per hectare, including clean-up costs. However, long duration floods (~2 months) are likely to kill improved grasslands, costing £200 per hectare a year for reseeding. An arable crop is likely to be killed after ~2–3 days of flooding and costs could be about £450–500 per hectare (Morris et al. 2004a). Assisted natural recovery should be considered when designing an offline storage area in order to create areas in a way that provides both biodiversity and FCERM benefits. To do this it is essential that biodiversity and geomorphology experts are involved in the design process.

One major issue that can elevate construction costs for a flood storage area is compliance with the Reservoirs Act. The current threshold above-ground storage volume for the full requirements of the act to become mandatory could change in the future from 25,000m³ to 10,000m³, and may include consideration of a cascade of smaller ponds on a floodplain as well as one large individual feature (Wilkinson et al. 2013a). However, these larger scales of storage measure quickly move from being an NFM measure to a traditional engineered one. More information on the implications for NFM and reservoirs legislation can be found in Wilkinson et al. (2013a), but further research is required on this subject area.



Figure 2.20 Offline storage pond located on a floodplain buffer zone in the Belford catchment, Northumberland, during a flood event (top panel) and after a flood event (bottom panel)

2.5.8 Research gaps

The literature has suggested there are a number of research gaps that still need to be addressed to fully understand not only how offline storage areas alter hydrological processes during floods, but also how they should be designed and maintained. Known gaps are summarised below.

- Further work is required to upscale the findings behind smaller scale offline storage areas. For example, assessing the impact of using many smaller scale offline storage measures such as RAFs and small washlands (like as those used at Holnicote) distributed across large catchment areas.
- When empirical evidence becomes available, models should be informed by these new datasets.
- There are large gaps behind designing offline storage areas to comply with the Reservoirs Act. Does a cascade of smaller ponds fall under this act and if so, what are the design criteria for the cascade?
- How do these types of feature function on groundwater fed catchments? Can they increase flood risk?
- Can these types of feature be designed to cover more multiple benefits, for example, improving low flows?
- How do we separate traditionally engineered flood storage areas from more naturally functioning storage area? What criteria should be used?
- Who maintains these features? How does maintenance vary across these measures? How often is it needed and how much does this maintenance cost?

Chapter 3. Woodland management





3.2 Catchment woodland



3.3 Cross-slope woodland



3.4 Riparian woodland



3.5 Floodplain woodland

Woodland management

This chapter forms the literature review for woodland WWNP interventions.

Sections 3.2 to 3.5 cover in detail each of the different WWNP interventions reviewed:

- 3.2 Catchment woodland
- 3.3 Cross-slope woodland
- 3.4 Riparian woodland
- 3.5 Floodplain woodland

3.1 Introduction

3.1.1 What is WWNP in the context of woodland management?

This section reviews the evidence concerning the role of woodland management in managing flood risk. It considers 4 woodland elements linked to placement within the landscape:

- catchment woodland
- cross-slope woodland
- riparian woodland
- floodplain woodland

Both the inherent role of the woodland in affecting flood risk and the impact of woodland management practices are considered.

Catchment woodland is defined as the total area of all woodland within a catchment, comprising general woodland cover of all types and species, including plantations, plus specific forms where present, such as cross-slope, riparian and floodplain woodland.

Cross-slope woodland is the placement of smaller areas or typically belts of woodland across hill slopes and includes all woodland types and species. It can be managed as either productive or unproductive woodland.

Riparian woodland is woodland located within the riparian zone, defined as the land immediately adjoining a river channel and influenced by it. This zone is usually relatively narrow, often extending <5 m on either side of watercourses, and typically comprises native broadleaved woodland that is often unmanaged. In the past, conifer plantations extended into riparian zones but most of these areas have now been cleared and being restored to native woodland.

Floodplain woodland comprises all woodland lying within the fluvial floodplain that is subject to a regular or natural flooding regime. It typically comprises broadleaved woodland and can range from productive woodland on drier parts to unmanaged, native wet woodland in wetter areas.

The 4 woodland elements are considered in turn and similarly structured in terms of the issues posed.

The term 'woodland' is used to describe land predominantly covered in trees (with a canopy cover of at least 20%), whether in large tracts (generally called forests) or smaller areas known by a variety of terms (including woods, copses, spinneys or shelterbelts). These terms are not specific to individual woodland types or species, although forests are often taken to mean conifer plantations, while small woods are generally assumed to comprise native broadleaves.

Woodland and forest terms are used interchangeably throughout the literature review and where relevant, type and species are specified. The biophysical processes whereby woodlands interact with flood generation, conveyance and thereby flood risk management are common to the 4 woodland elements described above, although vary in relative degree.

3.2 Catchment woodland

3.2.1 Understanding the science

Woodland has long been associated with an ability to reduce flood flows, with legislation enacted in the 19th century in both Europe and the USA to conserve forest cover for its water control and related functions (Anderson et al. 1976, McCulloch and Robinson 1993). Controversy over the extent of the forest effect led to the establishment of the first catchment studies in 1900 in the Bernese Emmental region of Switzerland and, in 1911, in the White Mountains of New Hampshire, USA. Early catchment observations supported the ability of forests to regulate stream flows, and further experimental catchment studies were set up to quantify the impacts on water yield, floods and erosion.

Legislation passed in 1936 in the USA gave the US Forest Service responsibility for conducting flood control surveys of forested catchments to determine management measures for 'run-off and water flow retardation'. This led to a marked expansion in the number of research studies and by the 1960s around 150 forested experimental catchments were in place across all regions of the USA. The vast majority of these studies looked at the hydrological effects of forest harvesting, replanting and road construction. Interest in the role of forests in streamflow regulation extended around the world, with many more catchment studies established across Europe, Australia, New Zealand and Africa.

The results of these studies have been the subject of multiple reviews (for example, Anderson et al. 1976, Bosch and Hewlett 1982, Best et al. 2003). Common findings are as follows.

- Forest felling increases peak flows.
- Forest planting reduces peak flows.
- The overall effects of felling or planting conifer forest on peak flows tend to be greater than those for broadleaves.
- Effects on peak flows tend to be less in the dormant season, especially for broadleaves.
- The effects of forest felling and planting on peak flows are greatest for small and medium flood peaks, declining in percentage terms with increasing flood size.
- Changes in peak flow magnitude due to forest cover tend to decline with increasing annual precipitation.

- The assessment of the impact of forest felling on large flood peaks is heavily constrained by the relatively short length of post felling data, dictated by the rate of forest regrowth.
- The effects of forest felling on peak flows decline over time and can be shortlived, depending on the rate of forest regrowth.
- It is very difficult to detect changes to streamflow when the extent of forest felling or planting is less than 15–20% of a catchment.
- It is very difficult to detect the response of streamflow to forest felling or planting in large catchments (>100km²) due to: the limited scale/area of change in forest cover usually involved; opposing effects due to other changes in land use and management in other parts of the catchment; and the increasing importance of river channel processes controlling flood flows downstream.
- Forest management practices such as cultivation, drainage and road construction can increase peak flows, depending on the scale, location, design and nature of practice, including use of good practice measures.
- Forest felling can increase soil erosion and sediment delivery, increasing flood risk by downstream siltation, although this can be successfully controlled by good management practices.
- Well-designed and managed forest planting can reduce soil erosion and sediment delivery, contributing to reduced flood risk.
- Forests can reduce the risk of flooding for some but not all flood events.

Much of the research reviewed by these papers involved well-designed before and after forest felling plus control, paired catchment studies (Before–After Control–Impact design, BACI), supporting robust statistical analyses. These findings are considered in more detail below.

3.2.2 Measurement of the impacts of forest felling

Among the best designed earlier studies were the well-known forest catchment clearfelling experiments at the Coweeta Hydrologic Laboratory in North Carolina and at the Hubbard Brook Experimental Forest in New Hampshire, USA. An analysis of all major storm hydrographs before (18 year baseline) and after (3 years) the complete clearfell to waste of a 44ha mature hardwood forest in the Coweeta basin increased quick flow volume by 11% (6mm) and peak discharge by 7% (Hewlett and Helvey 1970). Quick flow increases ranged from 0% for small floods to 22% for a record 7-day flood sequence. The increases occurred for storms in all seasons of the year and were thought to reflect the extended nature of soil moisture deficits. There was no effect on time to peak, recession time or quick flow duration. The absence of timber harvesting allowed the authors to conclude that the increase in flood peaks and volume was due to the removal of the forest water use/evaporation effect, rather than the result of soil damage/compaction or road construction.

At Hubbard Brook, the impact of clearfelling 86% of a 16ha mature hardwood catchment after an 8-year baseline period was studied by Hornbeck (1973). Herbicide was applied for 3 years after felling to prevent forest regrowth and thus to maintain the clearfelling contribution. Felling had a range of effects on growing season storms, ranging from a minor decrease to a 30mm increase in quick flow. The largest effect was on the most extreme event, where the daily peak flow rate was 178mm higher after felling. This was related to the difference in soil moisture deficit at the time, which was much greater before felling due to forest water use. Snowmelt was also more rapid, increasing most spring storm events by a maximum of
50mm. Storm events during the autumn and winter were unaffected; this was ascribed to the fact that soils had rewetted and tree water use was at a minimum. Time to peak and the duration of storm peaks increased following felling, but not by a statistically significant margin due to the high standard errors for the calibration period.

Another well-known US study involved the H.J. Andrews Experimental Forest and related sites in the western Cascades of Oregon. Studies of long-term records following 25–100% felling of 3 small catchments (60–101ha) nested within 6 large basins (62–640km²) found peak flows to increase by 13–16% for a 100% annual exceedance probability (AEP) flood and by 6–9% for a 20% AEP event (Beschta et al. 2000). Increases were larger for smaller, sub-annual peak discharges (<0.28-year return period), ranging from 31% to 116% in most basins (Jones 2000). No statistically significant increases were found for larger events, in a large part due to the small number of such recorded peak flows (for example, only one 2% AEP flood was recorded) and the measurement errors involved. Similarly, the authors did not find strong evidence for peak flow increases in the larger number of variables involved. Snowpack dynamics, cloud water interception (capture of cloud water by the forest canopy) and forest road density were significant factors explaining the variation in felling impacts between catchments.

A number of key studies on the impact of forest clearfelling on flood peaks have been undertaken in South America. Iroumé et al. (2006) analysed the effect of an 80% clearfell of the conifer covered 35ha La Reina catchment in southern Chile by storm size and found small peak flow events (5–10mm) to increase by 189%, medium events (10–50mm) by 74% and large events (>50mm) by 62%. Birkinshaw et al. (2011) updated these analyses in an integrated field and modelling (SHETRAN) study that used 1,000 years of weather generated rainfall data to look at the relationship between the effect of forest cover and increasing event size. The simulations showed that the absolute difference in discharge between forested and logged states remained approximately constant with increasing discharge but decreased as a percentage of discharge. Relative convergence appeared to start as flood frequency reduced below a 10% AEP flood for shallow soil conditions, but did not converge for deep soils. Nevertheless, forest cover was predicted to still reduce extreme flood peaks (for example, 1% AEP) by a sizeable margin (30–40%). Bathurst et al. (2011) also used a long (1,000 year) time series of synthetic climate data to extend the range of flood return periods to test the hypothesis that the effect of forest cover decreases with increasing peak discharge. In general, they found that as peak discharge increased to extreme levels, the difference between forest and non-forest scenarios decreased either absolutely or relatively. Nevertheless, it was concluded that forest cover could still significantly reduce damaging, moderate flood peaks.

Many other BACI designed catchment-scale forest felling studies are distributed around the world. Guillemette et al. (2005) reviewed the results of 50 of these from across boreal and temperate regions and found changes to what they defined as 'bankfull' peak flow (floods measured by individual studies as ranging from 1% to 67% AEP) to range from 0 to +170% in 49 of the 50 studies. A sizeable component of the variation in the results could be attributed to the percentage of the catchment felled, with the greatest increases in peak flows associated with >70% felling. However, the impact of 100% felling treatments ranged from zero to almost +170%, influenced by factors such as

- the nature of the climate, particularly dryness and amount of snowfall
- peak size
- altitude
- forest type

- forest age
- nature of felling practice
- presence of forest roads

These findings contrast with those from the 2 BACI conifer forest harvesting catchment studies at Plynlimon in Wales and Balquhidder in mid Scotland. Although forest growth appeared to reduce peak flows at Plynlimon, the impact of felling between 26% and 50% of 3 subcatchments (90–370ha) and 32% of the upper catchment of the main River Severn (8.7km²) had no detectable effect (Robinson and Dupeyrat 2005). Similarly, felling of 40% of the Kirkton Glen catchment at Balquidder did not result in an increase in peak flows (Johnson 1995). The difference in behaviour between UK and overseas studies is not thought to be due to climate or physiographic differences (UK study conditions fall within the overseas range), but to the restricted scale and/or the extended time period over which the felling was carried out in the former cases, with a maximum of 20% of a catchment felled in any one year and felling extending over a period of between 5 and 17 years. It is known that the effects of clearfelling decline over time due to the rapid regrowth of restocked crops (Lin and Wei 2008, Iroumé et al. 2010).

3.2.3 Regional scale studies

A number of national and European studies have used regional flow datasets to investigate the effects of forestry and land cover in general on flood flows and frequency. L'Ubomír et al. (2011) found a positive relationship between land cover changes associated with accelerating direct run-off (including deforestation) between 1990 and 2006, and increasing flood frequency using flow data from a number of small catchments in Slovakia, which was clearest for catchments with very high flood potential. In contrast, Gustard et al. (1989) could not find a statistical significant relationship between the proportion of forest cover and the mean annual flood in their assessment of catchment flow data from over 40 European agencies. Similarly, assessments of historical flow data from the UK hydrometric network have struggled to detect an effect of forestry on different flood parameters (NERC 1975, Beven et al. 2008). In general, variability in weather conditions between years and inconsistencies in rainfall and flow data dominated any tendency for changes over time. Variable portions of forest cover between catchments and concurrent changes in land management within non-forested areas also had an impact on the ability to identify trends.

In their assessment of hydrological results from 28 research basins across Europe, Robinson et al. (2003) found that forestry could have a significant effect on peak flows (taken as the 5 highest flows in a year) at a local level but not at a broader regional or European scale. At Chiemsee in southern Germany, conifer planting on farmland reduced peak flows by around 100% by the time trees reached 20 years of age, while in the other studied sites reductions of 10–20% was more typical. Pre-planting drainage operations had the opposite effect of increasing peak flows by 15–20% for 15–20 years, while clearfelling could increase flows by around 10%, although this was often difficult to detect. Clearcutting eucalyptus plantations increased peak flows by about 50% but the effect only lasted for 1–2 years due to rapid regrowth of the crop. Overall, it was thought that these local effects could easily be lost at a larger basin scale due to spatial dilution or being cancelled out by the contrasting effects of the different forest management or other land use activities.

The findings of these regional studies contrast with a more recent Europe-wide assessment by the European Environment Agency of the water retention potential (in terms of water absorption or use) of forest cover (EEA 2015). Four hydrological indicators (run-off coefficient, surface run-off coefficient, run-off irregularity coefficient and flushing ratio) were calculated for 287 sub-basins comprising >65,000 catchments and used to estimate the water retention potential according to forest cover (measured in hectares), forest type (conifer, broadleaved and mixed) and degree of management (protected and unprotected/commercial). Key findings based on a computed index of the coefficient values were as follows.

- Water retention was 25% higher in water basins with 30% cover and 50% higher in those with 70% cover, compared with basins with 10% cover.
- Water retention was typically 25% greater in summer than in winter.
- Basins with conifer forests generally retained 10% more water than those with broadleaved or mixed forests.

No clear conclusions could be drawn regarding the impact of the degree of forest management. In general, forests in Alpine and continental regions provided the highest water retention potentials. However, there was significant variation in the regional relationships, indicating that much depended on local conditions (water retention potentials were also high in lowland areas of Atlantic and Boreal regions).

3.2.4 Modelling studies

In addition to the research catchment studies, a range of modelling studies have drawn on process understanding to simulate the effects of catchment woodland on flood flows. Caution is required when considering these results as they rely heavily on the ability of modellers to accurately parameterise relevant processes and upscale these from the plot to the catchment level.

Notable studies include an application of 2 different models at Pontbren in mid Wales. The first involved a physics-based model and predicted that the 100% afforestation with deciduous trees of the 12km² headwater catchment would reduce an extreme flood peak (140mm of rainfall over 2 days, with an estimated return period of 180 years) by an average of 36% (with 95% confidence intervals of a 10% and 54% reduction) (McIntyre et al. 2012). A second run-off generating model used regionalised values of flow indices from national datasets (based on Hydrology of Soil Types) and predicted that full afforestation would reduce a 10% APE flood by 12–15% (Bulygina et al. 2009). The latter results were considered less reliable than the former due to use of national scale generalisations, rather than local knowledge and data, as was the case with the first model.

A different modelling approach was followed by Thomas and Nisbet (2016), who combined the Hydraulic Modelling System (HEC-HMS) and the USA Soil Conservation Service Run-off Curve Number method to simulate the effect of catchment woodland on flood flows in the Pickering Beck catchment in north Yorkshire. The method predicted that converting the existing 25% woodland cover to improved grassland would increase the peak flow for a 1% AEP flood by 41%. A similar result was obtained by the application of the same modelling approach to the Taihu Lake catchment in China. Rongroung and Guishan (2007) predicted that converting land cover from forest to grassland would increase flood peaks by 45% and reduce time to peak by 5 hours.

In France, Cognard-Plancq et al. (2001) applied a rainfall–run-off model (GRHUM) to stream data from the Mont Lozère experimental catchments and concluded that forest cover could reduce annual daily flood peaks (>33% AEP) by up to 20% and flood volumes by 10%. In the Pacific Northwest, Storck et al. (1998) applied a GIS-based Distributed Hydrology Soil Vegetation Model (DHSVM) to 3 logged catchments and predicted that forest harvesting would increase peak flows by between 10% and 31%, depending on event size, amounts of antecedent snowfall, scale of harvesting and presence of forest roads. Wahren et al. (2012) used a spatially distributed rainfall–run-off model (AKWA-M) to investigate the impact of 3 afforestation scenarios (ranging from 35–99% cover) on summer floods in the Schlettenbach

experimental catchment in Saxony, Germany, and found small to medium sized events to be reduced by 3–70%, depending on pre-event catchment wetness.

3.2.5 Assessment of impacts of felling on large floods

Efforts at examining the impact of forest felling on large or more extreme flood events have tended to show reducing or smaller increases in peak flows (expressed in percentage) that are often not statistically significant (Beschta et al. 2000, Jones 2000, Birkinshaw et al. 2011). A major problem with such studies is the relatively short period of time before the regrowth of the felled site re-establishes the forest effect, which severely limits the scope for capturing more extreme events to determine the impact of felling. Replanted or regenerating trees often grow quickly utilising nutrients released from felling residues, becoming largely established within a period of 5–10 years. Kuraś et al. (2012) noted that the characteristically short period of record (often <10 years) in paired catchment studies is inadequate to statistically test changes in peak flows with return periods greater than 10 years. Another criticism is the failure of studies to account for possible changes in return interval between pre and post felling periods (Alila et al. 2009).

While attempts have been made to circumvent the first of the above problems by utilising flow records to calibrate hydrological models and then applying these to weather generated, longer term datasets (see, for example, Bathurst et al. 2011, Birkinshaw et al. 2011), none of these studies have addressed the return interval issue. A different approach was adopted by Kuraś et al. (2012), who used the physically based, spatially distributed hydrologic model DHSVM and synthetic climate data to take into account pre and post forest felling changes in flood frequency. In contrast to other studies, they found the effects of felling to increase with return period, which was attributed to the nature of the peak flow run-off generating processes in their snow-dominated catchments in British Columbia. Felling of 20–30% of the conifer cover had no significant effect on the peak flow regime, but a 50% clearfell was found to increase peak flows by 9–25% for 1–10% AEP floods. Peak flow frequency increased with return period after harvesting, with the largest 1% AEP event becoming 5–6.7 times more frequent.

3.2.6 Measurement of the impact of new planting

Due to the length of time it takes for forests to become established, only a small number of catchment studies have measured the impact of new planting on flood flows. A BACI designed small (310ha) catchment study by Fahey and Jackson (1997) in New Zealand investigated the impact of 67% afforestation by Radiata pine on previous tussock grassland. After 10–12 years of tree growth, mean flood peaks had fallen by between 55% and 65% across 3 peak size classes, while quickflows had decreased by 45–55%.

Closer to home, the only UK example is the long-term, upland forest hydrology catchment study at Coalburn in the north of England. This is the subject of the key case study cited below, which suggests that 90% afforestation by Sitka spruce of the 150ha catchment in 1972 produced a 5–20% reduction in peak flows, declining with increasing peak size (Birkinshaw et al. 2014). A shift in flood frequency was also noted, with an event of a return period of 13 years reducing in frequency to a return period of 20 years. Overall, flood frequency reduced by ~50% across all events, although care is required in interpreting these results due to the highly nonlinear relation between flood frequency and magnitude.

Archer (2003) examined how the impacts at Coalburn translated downstream to the much larger, 19% afforested, River Irthing catchment (335km²). He found annual pulse/peak flow numbers reduced by nearly 40% and pulse duration increased by >20% at Coalburn compared with conditions under the original moorland cover, while there was little evidence of change to these flow indices in the River Irthing catchment until the late 1980s.

Thereafter, pulse numbers reduced and duration increased in line with Coalburn and the much smaller scale of change.

3.2.7 Effect of catchment size on forest impact

Nearly all woodland felling and new planting studies have focused on small catchments (<10km²), reflecting the increasing difficulty of measuring flows, controlling land use change and ensuring watertight conditions as catchment size increases. Very few studies have examined the impact of forestry in very large catchments (>1,000km²) and those that have display less consistent findings, largely due to the problem of separating the effects of background changes, including trends in annual rainfall (Ranzi et al. 2002, Zhang et al. 2012).

3.2.8 Consideration of underlying processes driving forest effect

Much of our understanding of how trees and their management affect the generation and conveyance of flood flows is derived from studies of hydrological and soil processes. Key processes are considered to be:

- water use by trees
- related effects on snow accumulation and melting
- soil infiltration beneath woodland
- surface or hydraulic roughness exerted by woodland
- impact of woodland on soil erosion and sediment delivery

The role of each of these is considered below.

Water use

Much of the measured effect of forests on peak flows is assigned to the ability of trees to use more water than shorter forms of vegetation. The higher evapotranspiration rates of forest vegetation, particularly due to canopy interception but also to deeper rooting in some cases, is well-established in tree physiology, micrometeorology and catchment studies (Calder 1990, Calder et al. 2003, Nisbet 2005).

Several worldwide reviews of hundreds of catchment-based water balance studies have shown annual water use to be typically 400mm greater for a complete conifer cover and 200–250mm for broadleaves, compared with grassland, for annual rainfall exceeding 1,500mm (Bosch and Hewlett 1982, Zhang et al. 2001, Vertessy et al. 2003). The forest effect reduces with declining annual rainfall and is difficult to detect in very dry climates where annual rainfall is <500mm. Conifer forests generally lose between 25% and 45% of gross rainfall by interception, while the figures for broadleaves tend to range between 10% and 25% (interception loss for grass is close to zero, while that for other vegetation covers such as heather and bracken can come close to that of broadleaves) (Nisbet 2005).

Interception can produce a smaller but significant reduction in storm rainfall (1–2mm per day for broadleaves and up to 7–8 mm per day for conifers (Rothacher 1963, Calder 2003), which declines as a proportion of rainfall with increasing storm size. Iroumé and Huber (2002) showed that high daily interception losses can take place throughout the year, with canopy losses of up to 1mm per hour recorded in winter periods.

More important is the effect of the drier and better soil conditions that develop under woodland cover during the growing season. The greater water use by trees over consecutive days leads to a larger soil water deficit in woodland soils, which reaches a peak in summer periods. This can amount to tens or even hundreds of millimetres of potential additional storage, depending on climate, soil depth and forest type (Calder et al. 2003). Measurements of different soil types by Harrold et al. (1962) in a catchment study at Coshocton, Ohio, USA, found non-capillary pore space in the upper 20 cm depth to be 15–55% greater under forest compared with 'idle' land. Wahren et al. (2012) compared available soil water capacity between arable and 60 year-old forest plots in Saxony, Germany, and found soil porosity to be 22% higher in the forest topsoil, equating to 66mm additional soil water storage.

Soil water storage capacity is generally greater in drier and warmer areas, where it can result in a soil moisture deficit being carried over between years. The more available soil storage, the greater the potential for reducing storm run-off, although once filled, will not regenerate until drawn down again by woodland water use in the next growing season. The build-up of organic matter in woodland soils and the development of a litter layer also increase the water storage capacity (Feustel and Byers 1936). Tsukamoto (1975) found that the removal of forest litter increased peak flows by 168% for <100mm storms.

Snow

The ability of trees to affect snow accumulation and the rate of melting is another significant factor, especially for snowmelt-driven spring flood events (Anderson et al. 1976). Snow accumulations are generally lower beneath forest canopies due to aerodynamic factors and evaporation losses from snow cover on tree canopies (Calder 1990). The melting of snow on the forest floor can be significantly delayed by canopy shade, helping to desynchronise contributions from different parts of a catchment (Goodell 1959). In studies of rain-on-snow floods in Oregon and California, Anderson (1969, 1970) estimated that conifer shading reduced the melt rate by 40% (and flood flows as a consequence by 10%).

Soil infiltration

Another important process is soil infiltration, with higher infiltration rates generally associated with woodland soils, especially compared with those under agriculture impacted by livestock grazing or arable cropping (Bracken and Croke 2007). Tree cover protects soils from physical disturbance and, together with leaf fall and tree rooting, helps build up soil organic matter and creates good soil structure (Mapa 1995). The development of an interconnected system of soil pores or 'macroporosity' under trees encourages rainwater to enter the soil and follow deeper pathways to streams (Pritchett and Fisher 1987, Neary et al. 2009).

Studies have found woodland soils to be characterised by high infiltration rates usually in the order of hundreds or even thousands of mm per hour (Archer et al. 2013), which are rarely exceeded by rainfall intensity and thus much less likely to generate infiltration-excess overland flow (Anderson et al. 1976, Carroll et al. 2004a). A number of researchers have looked at the related measurement of soil-saturated hydraulic conductivity and shown ratios for woodland versus grassland values to range from 2 to 140 (Chandler and Chappell 2008, Alvarenga et al. 2011).

While the infiltration effect is lost as poorly or imperfectly drained soils become saturated following prolonged wet weather (often a characteristic of extreme winter floods), it can help to slow the generation of floodwaters from other soil types and at other times of the year. A notable example are more freely draining soils on moderate or steep slopes that have typically developed a compact surface turf as a result of many years of sheep trampling and

grazing. Such compaction can lead to rapid surface saturation and run-off, leaving deeper soils relatively dry beneath.

Studies at Pontbren in Wales have shown that woodland planting can quickly disrupt surface compaction, allowing more rainfall to infiltrate to depth and increasing the available soil water storage capacity. Soil infiltration rates were found to be 67 times higher within young native woodland shelterbelts compared with adjacent grazed pasture soils (Marshall et al. 2014). Wahren et al. (2009) measured soil hydraulic conditions in comparable soils under arable, 6 year-old afforestation, 50 year-old afforestation and ancient natural forest at a site in Germany. They found values to be 3–4 times higher under 50 year-old afforestation and ancient natural forest, and around 2 times higher for young afforestation compared with arable. These soil changes were incorporated into a rainfall–run-off model and were predicted to reduce peak flows for a 4% AEP flood by 7–11% and for a 1% AEP event by 4%.

Surface roughness

Surface roughness is usually greater under woodland than other vegetation types, which helps to slow the rate of surface run-off across the land. Tree butts, surface roots, deadwood and leaf litter all contribute to roughness and exert a barrier or drag effect on surface flows. Chow (1959) showed that the hydraulic roughness associated with a dense stand of willow coppice on the floodplain can be >5 times that of grass. However, this effect can be short-circuited by linear cultivation and drainage channels, as well as by forest roads and tracks, which concentrate and direct run-off along smoother and faster pathways. For example, large-scale deep ploughing and drainage affecting 90% of the Coalburn catchment was found to increase peak flows by 20% and reduce the time to peak by a third (Robinson et al. 1998). The impact was greatest on smaller peak flows and appeared to be lost for annual maxima, which was thought to reflect the wet, peaty nature of the catchment's soils and the greater extent of soil saturation and overland flow during larger events. Forest roads have similarly been found to increase peak flows, especially where there is a dense network (Anderson et al. 1976).

Sediment delivery

Sediment delivery to watercourses is increasingly viewed as an important factor in flood risk management (McIntyre and Thorne 2013). Downstream siltation reduces flood conveyance and increases the risk of local flooding, leading to demand for more dredging with consequent environmental problems. Well-managed woodland is generally associated with much lower sediment losses compared with other land use activities (Liu et al. 2005, Collins and Walling 2007, Vasquez-Menandez et al. 2010), contributing to flood alleviation. This reflects the ability of tree cover to protect soils, slopes and river banks from disturbance, as well as improving soil structure and increasing strength through organic matter inputs, tree rooting, soil drying and reduced surface run-off (Benito et al. 2003, Nisbet et al. 2011b). As a result, woodland forms the preferred land cover for catchment protection in many parts of the world.

On the other hand, poorly managed woodland can diminish or reverse this protective function, particularly when associated with cultivation, drainage, road construction and harvesting practices that can increase soil erosion and sedimentation.

3.2.9 Confidence in the science

There is high confidence in the understanding of woodland hydrological processes and the measured and modelled data demonstrating that catchment woodland can reduce surface

run-off and small to medium peak/flood flows in small catchments (<10km²). While much of the measured data are derived from international studies, the general consistency of the findings and the fact that many of the studies involve sites with similar climatic and physiography conditions to the UK increases confidence in the results. The level of confidence declines with increasing peak and catchment size, and remains particularly disputed for flood peaks greater than a 1% AEP flood and catchment areas >100km².

The lack of confidence for large flood peaks stems mainly from the absence of measured data due to the great difficulty of determining changes for relatively rare events. There are good modelled data indicating that catchment woodland could reduce large events, especially summer floods, due to higher soil moisture deficits under woodland (particularly conifer). However, process understanding indicates that there are limits to some processes, becoming less effective with increasing event size. This mainly applies to the water use and soil infiltration benefits of woodland, with daily interception losses from conifer forest predicted to reach a maximum of 7mm for daily rainfall totals greater than 70mm, while available soil water storage will increasingly be filled by high winter rainfall (depending on soil depth and type), with soil infiltration benefits disappearing once soils become fully saturated.

Process understanding and modelling suggest that there is no catchment size/area threshold for catchment woodland to reduce flood flows, although the effect is likely to decline with the increasing importance of river channel processes downstream in larger catchments. The main issues with larger catchments (>100km²) are:

- the tendency for the proportion of forest cover, and therefore its contribution, to decline
- the greater the chance that other changes to land use and/or management in the wider catchment will act to 'swamp' or offset the forest effect (for example, general trends of urban expansion/development and agricultural intensification)
- the increasing scope for tributary synchronisation and desynchronisation effects to moderate downstream impacts

From an experimental point of view, it is almost impossible to maintain a large control catchment over time, while errors linked to flow measurements will increase due to the difficulty of constraining high flows within flow gauging structures. A particular challenge for modelling studies is the scaling up of process understanding from the plot/site to large catchment level.

In contrast to the scientific understanding, the level of confidence shown by stakeholders in the ability of catchment woodland to reduce flood flows varies from low to medium (based on personal experience). This arises for a range of reasons, including:

- mixed views on the science, partly driven by mixed measured and modelled results from UK studies
- a tendency to focus on large floods in larger catchments, for which the data are less certain
- the lack of a clear number for the woodland contribution and concern about the number of variables influencing its effectiveness
- concern that catchment woodland could increase flood risk due to synchronisation effects and in particular, the washout of woody debris
- · a dislike of conifers and the weaker contribution of broadleaves

 the absence of easy to use, reliable and robust modelling tools to predict the woodland effect

3.2.10 Metrics of cost–benefit and multiple benefits

There appear to have been few attempts at quantifying the cost–benefit of catchment woodland for flood risk management. A major problem is the catchment-specific nature of the assessment and the detailed modelling and data required. This is hampered by the lack of:

- robust models incorporating the main woodland processes with validated parameter sets
- assessments of how woodland effects interact with existing flood defence measures
- methods for translating the woodland contribution into an economic value

Nisbet et al. (2015) calculated the benefit–cost ratio of catchment woodland for flood risk management as part of the Slowing the Flow at Pickering project; however, the numbers combined the effect of different woodland measures (woodland creation plus the installation of LWD dams and timber bunds) with moorland and agricultural measures. The benefit–cost ratios for the set of woodland only measures were estimated to range from 1.5 to 3.0 for flood regulation (for reducing the chance of flooding from 25% to <4% in any given year).

More recently, Smithers et al. (2016) estimated the value of flood regulation services provided by existing woodlands for inclusion in the UK ecosystem accounts. A very simple and weak approach was adopted due to the lack of available data for a national application, which relied largely on extrapolating the model predictions for the impact of woodland cover on peak flows from the Pontbren study in Wales (see Case Study). National spend figures on flood defence were assigned on a catchment basis and used to calculate the value of woodland for flood regulation based on a replacement cost method. The asset values of existing woodland cover in relation to the notional annual reduction in fluvial flood expenditure were estimated at £1.3 million to £1.5 million.

The wider, multiple benefits of catchment woodland are well known from studies over the past 20 years. Work on calculating an economic value for these is most developed for climate regulation (carbon sequestration), timber provision and recreation. The UK National Ecosystem Services Assessment (2014) estimated the costs and benefits of different forest planting schemes across the UK. It found that the inclusion of greenhouse gases and recreation benefits shifted the net cost/benefit from -£65 million per year based on maximising market/timber value to +£546 million per year. This makes it clear that the incorporation of non-market values into decision-making over where to plant forests provides significant gains for society as a whole.

Forest Enterprise England has published Natural Capital Accounts for the public forest estate in England for 2015 to 2016 (Forestry Commission England, 2016). The annual value of the services delivered by England's woods and forests included timber provision (£7 million), climate regulation (£83 million), recreation and public access (£148 million), plant and seed supply (£4 million) and minerals (£0.6 million), giving an annual total of £243 million. The net asset value over a 50-year time horizon was estimated at £11.9 billion.

On an individual catchment basis, Nisbet et al. (2015) valued the ecosystem services provided by the range of NFM measures implemented as part of the Slowing the Flow at Pickering project. This valued the contribution of the woodland measures in terms of habitat creation, climate regulation, erosion regulation and flood regulation. As noted earlier, only a combined value was calculated for the woodland measures, which give a benefit–cost ratio

of 5.6:1. Climate regulation contributed the greatest value, followed by flood regulation and habitat creation.

3.2.11 Effectiveness/performance

The effectiveness of catchment woodland for reducing flood risk is influenced by a large number of factors including climate, soil and geology, woodland type, woodland and catchment scale, measurement errors, woodland management and design, and the nature of the alternative land use. These are considered in turn below.

Climate

The nature of the climate (especially the amount and distribution of rainfall, as well as evaporation demand) affects woodland water use and therefore the magnitude of the soil water deficit and thus potential floodwater storage below ground. Summer soil water deficits typically range from 25mm to 50mm in the north-west to several hundred mm in south-east England (Met Office 2016). Woodland water use could enhance these values by 50mm or more, with greater scope for the additional deficit to persist through the winter in the drier south and east (Calder et al. 2003). The drier the soil before a flood event, the more water retention and potential to reduce flood flows. The amount of snow is another important factor as woodland can both reduce the accumulation of snow (via the evaporation/ interception of snow on the canopy) and the rate of melting (Anderson et al. 1976). This will impact most on spring melt flood events. As noted earlier under 'Water use' in Section 3.2.7, rainfall duration and intensity are key factors affecting forest interception loss (Calder et al. 2003).

Soil and geology

Soil type and depth influence the soil water storage capacity and the availability of water to sustain woodland water use during dry periods, and thereby the size of the woodland effect. Soil type also determines soil vulnerability to damage and thus the relative size and significance of the soil infiltration benefit. Geology exerts a strong control over run-off pathways and the ability of catchment woodland to affect these. The more porous the geology, the less scope for woodland processes to affect surface run-off, particularly by enhanced infiltration and hydraulic roughness.

Woodland type

Woodland water use is strongly affected by woodland type, being greatest for conifers. Conifer interception losses are typically twice those of broadleaves and more than 3 times for maximum daily interception loss in summer and 7 times in winter (Calder et al. 2003). This results in higher and more sustained soil moisture deficits under conifer, with a greater capacity to reduce flood flows. There are no catchment studies comparing the effects of conifers with broadleaves on flood flows, but clearfelling studies generally show larger percentage increases in peak flows for conifer catchments (Anderson et al. 1976).

Soil infiltration rates can be lower for conifers (Archer et al. 2013), but this can reflect the impact of poor practices associated with more intensive management of plantation forests or younger aged stands. Well-managed woodlands, irrespective of forest type, exhibit high infiltration rates, with studies showing rates to be highest under old-aged stands of both conifer and broadleaves (Archer et al. 2013). The potential high water use of willow and poplar species, especially when well supplied with water such as in riparian and floodplain habitats, can result in much higher soil water deficits and potential below-ground floodwater storage compared with other broadleaves (Finch et al. 2011). Hartwich et al. (2016) found

available water capacity in the soil to increase by 26% for short rotation coppice (SRC) compared to annual crops. SRC is also very good at rapidly establishing high hydraulic roughness to hold back and delay the propagation of floodwaters (Environment Agency 2015b; see Section 3.5.1).

Woodland and catchment scale

In general, the larger the extent of woodland cover, the greater the expected impact on flood flows. This simply reflects the footprint of the woodland at the catchment scale and thus the relative contribution of the different woodland processes. A related factor is that the smaller the proportion of woodland cover, the greater the predominance of non-woodland land use and management effects, as well as changes to these which can reinforce, overwhelm or offset the woodland effect (Jones 2000).

Woodland placement within catchments and catchment geometry/structure also has a role to play by influencing the relative timing of the woodland contribution to peak flow response. This has potential to be positive or negative by desynchronising or synchronising flows, respectively, from different parts of a catchment, particularly involving tributary contributions. In general, the more rapid the tributary response and the closer the location to the community or asset at risk of flooding, the greater the scope for the woodland delaying function to negate or reverse any flood storage effect (Odoni and Lane 2010).

Measurement errors

The size of errors associated with flow measurements affects the ability to detect a change in flood peak (Beck et al. 2013). Errors are smallest where a flow controlled structure is installed, and resulting water level measurements are converted to discharge using the design rating for the structure (checked in the field). This will typically give errors in the range of $\pm 2.5-5.0\%$, but will only apply while flows remain within the structure. With increasing flood size, it is more likely that the flood peak will come out of bank and bypass or overtop the structure, giving much higher errors. Similarly, larger errors apply to measurements made in uncontrolled/natural river channel sections with simple water level recorders, more typical of monitoring studies. Errors of at least $\pm 10\%$ can be expected, meaning that the scale of the woodland effect would need to exceed this to be measurable (Beschta et al. 2000). Thus, if woodland (that is, 100% cover) was able to reduce flood flows by 50% per se, the impact of woodland creation is unlikely to be detectable unless it involved at least a 20% change in catchment cover. This accords with the general finding from clearfelling studies that no effect can be detected when <20% of a catchment is felled (Bosch and Hewlett 1982, Cornish 1993, Stednick 1996).

The spatial threshold is expected to increase with the size of the flood peak in line with the difficulty of accurately rating flood flows, the propensity for the rating to be affected by physical changes to the river channel and floodplain conditions, and the likely declining forest effect. A combination of the increasing measurement error and decreasing number of recorded peak flows with increasing peak size makes it very difficult to statistically prove a change in peak height in response to woodland creation or management for large/extreme events, even where large changes in forest cover are involved (Kuraś et al. 2012). This partly explains the paucity of measured data and evidence at the large catchment scale. Failure to account for a rising trend in rainfall due to climate change can be another important source of error, especially where a control catchment is absent (Zhang et al. 2012).

Measurement errors for flood flows can be reduced through technological developments such as the use of acoustic Doppler current profiling by a remote controlled boat, although the challenge remains to be onsite at the right time to capture such conditions. In the future,

advances in flow sensors and the application of remote sensing may provide a more accurate way of measuring out of bank flows.

Woodland management and design

Woodland management practices associated with productive woodlands can increase peak flows and thus influence the overall contribution of catchment woodland. Cultivation and drainage, road construction and felling/harvesting operations have the greatest potential to impact on floodwater storage/evaporation and response times. However, much depends on the nature/standard and scale of practice, with the poorer and more extensive the operation. the larger the impact. Old style deep ploughing and drainage can speed up surface run-off, reducing time to peak and increasing peak height, although the effect declines with increasing peak size and can be difficult to detect for flows greater than the mean annual flood (thought to reflect the increasing predominance of saturated-excess overland flow) (Robinson et al. 1998, Archer and Newson, 2002). Developments in good practice such as the use of shallower forms of linear cultivation, gentler drain gradients and discharging cultivation channels and drains to buffer areas are expected to reduce the impact on peak flows, although there is a lack of both measured and modelled data to quantify this. Run-off from woodland track and road surfaces and associated road drains can also increase peak flows (Jones 2000), while good practice measures such as disconnecting road drainage from natural watercourses should help to mitigate this effect.

Clearfelling has potentially the biggest impact of all forestry practices by removing the tree cover, reducing water use and rewetting soils, while the soil compaction and rutting associated with poorly managed timber harvesting can greatly reduce soil infiltration and increase overland flow and sediment delivery to watercourses (Birkinshaw et al. 2011). These effects can be partly offset by the action of harvesting residues/brash, which can exert a significant interception loss (equivalent to heather; Nisbet 2005), increase surface roughness and help protect soil from ground damage (Nisbet 2001). However, the main way of controlling the impact of clearfelling is to restrict the scale of the activity at the catchment level. Since forest regrowth is generally rapid, the water use effect can be largely restored within a 1--15 year period, depending5 year fallow period to control weevil damage will extend the recovery period. Limiting the scale of clearfelling to <20% of the catchment upstream of at risk communities/assets in any 10-15 year period will minimise the impact and make it unlikely to be detectable (Bosch and Hewlett 1982, Cornish 1993, Stednick 1996).

Woodland design influences woodland water use and particularly hydraulic roughness. Water use is generally greatest for closed canopy woodland, although studies show that the introduction of some open space by thinning treatments has little effect until up to a third of the canopy is removed. This is supported by modelling studies and explained by the increased canopy ventilation/turbulence driving interception losses and offsetting the effect of the temporary decrease in canopy cover, which tends to rapidly infill (Teklehaimanot et al. 1991). Open canopy, low density woodland can be expected to intercept less water and have wetter soils compared with closed canopy woodland, reducing floodwater storage. It is also likely that lower density planting will delay the recovery of soil conditions on damaged sites, reflecting the slower development and reduced extent of tree root systems. However, a patchy distribution of soil improvement will help by intercepting run-off from more damaged adjacent areas.

Tree spacing has a direct effect on the degree of hydraulic roughness. Unpublished calculations using the simplifed parameter of Manning's n indicate that the contribution of the tree stem to restricting surface flows diminishes to a small value once trees are wider than 5 m. Hydraulic roughness increases sharply as tree spacing reduces from 2.5m to 1.0m, though this will be partly offset by the accompanying reduction in the amount of ground

vegetation and shrub layer due to shading. Tree diameter, the number of stems per tree, the amount of deadwood on the ground, and the size of tree butts and related microtopography, will also influence hydraulic roughness. Woodland age can have a significant influence on water use, with transpiration rates greatest for actively growing young stands and diminishing with old age (Harr 1986, Roberts et al. 2001a, Vertessy et al. 2001).

Nature of alternative land use

The type of land use being replaced by woodland and the way that it has been managed will influence the relative size of the woodland effect. This reflects differences in water use between other land cover types, which tends to increase in the order bracken > heather > grass > arable, as well as the propensity of associated land management practices to compact or poach the soil and increase surface run-off and sediment delivery to watercourses (Nisbet 2005). The lower is the water use, the more damaged the soil and the lower the hydraulic roughness of the baseline land use, the greater the net benefit of woodland creation for reducing flood flows.

3.2.12 Key case studies

A number of case studies have been established in the UK to assess the impact of catchment woodland on stream flows. The best designed are the 3 long-term forest hydrology catchment studies at Coalburn in northern England (Robinson et al. 1998), Plynlimon in mid Wales (Marc and Robinson 2007) and Balquhidder in mid Scotland (Johnson 1995). These were set up in the 1960s or 1970s as research studies to determine the water balance of upland conifer forest versus grassland. All included flow measuring structures to record changes to flows, rain gauge networks and automatic weather stations to quantify rainfall inputs and evaporation losses, and were supported by a wide range of process studies, including measurements of woodland transpiration and interception rates.

Of the 3 studies, **Coalburn** is most relevant to determining the effect of catchment woodland on flood flows. It was established as a before and after study of the effect of large-scale (90%) planting of an upland moorland catchment with predominantly Sitka spruce. Measurements started in 1967 and have continued to the present, allowing the contrasting hydrological effects of the different phases of a woodland cycle to be assessed. The main limitations of the study are that it does not include an equivalent monitored control catchment and the intensive nature of the forestry establishment practices, which included deep ploughing and drainage of the catchment's peaty soils. Details of the research work are included in the accompanying <u>Evidence Directory</u>.

In contrast, the **Plynlimon** and **Balquhidder** studies focused on measuring the impact of forest clearfelling. Both struggled to detect any significant changes to peak flows (Johnson 1995, Marc and Robinson 2007), which is likely to be due to the restricted scale and/or the extended time period over which the felling was carried out, with a maximum of 20% of a catchment felled in any one year. The Balquhidder study also included conifer planting in an adjacent catchment, but the assessment was similarly constrained by the small scale of land cover change (14% of catchment planted).

Other studies of catchment woodland in the UK are much more recent. There is an absence of catchment-scale hydrological measurements of broadleaved and lowland woodland, with all NFM type studies limited to more targeted, small-scale woodland creation less than 10 years old (see Sections 3.3, 3.4 and 3.4 on cross-slope, riparian and floodplain woodland respectively). Quantitative data are essentially limited to modelling studies that have considered larger scale or whole catchment woodland planting scenarios. These include model applications at Pontbren in Wales, the Hodder catchment in northern England and the Parrett catchment in south-west England (McIntyre and Thorne 2013). Unfortunately, it is

unclear which woodland processes were included in the different models and the values that were selected to represent them. None incorporated the effect of woodland creation on sediment interactions with flood flows.

Process measurements of woodland water use are available at the plot scale for lowland broadleaved (including SRC and short rotation forestry) and conifer woodland at a number of sites in England including at:

- Thetford Forest in east England (Roberts 1983)
- Clipstone in the East Midlands (Calder et al. 2002)
- Blackwood in Hampshire (Roberts et al. 2001b)
- Old Pond Close in Northamptonshire (Harding et al. 1992)
- Squerreys Estate in Kent

The results from these studies have been used to improve model performance.

The most comprehensive modelling assessment of the effects of catchment woodland on flood risk is probably the ongoing study at Southwell in Nottinghamshire (Dixon and Scott 2017). This involves the use of the TUFLOW model to represent water use, infiltration and surface roughness processes. Also included is an evaluation of the potential contribution of woodland planting to the economics of flood risk management. Further details on the case study are given in the <u>Evidence Directory</u>.

There are a large number of international case studies on the impact of catchment woodland on flood flows, but most involve assessments of clearfelling. The best example of a catchment scale (310ha) woodland planting study is at Glendhu on the South Island of New Zealand, which involves converting tussock grassland to a Radiata pine plantation (Fahey and Jackson 1997).

3.2.13 Funding

Funding for woodland creation is primarily through the Rural Development Programme (RDP) for individual countries as part of the European Agricultural Fund for Rural Development: Europe investing in rural areas. In England, this is administered by Natural England, Forestry Commission England, Defra and the Rural Payments Agency under Countryside Stewardship. Unlike the previous English Woodland Grant Scheme, which offered an 'Additional Contribution' of £2,000 per hectare for planting within priority areas to reduce flood risk, Countryside Stewardship is a purely points based system. It is a targeted and competitive scheme, with grants awarded to those who deliver the most for biodiversity, water (quality and flooding) and climate change.

Funding is provided to supply, plant, weed and protect trees as a one-off capital payment. The maximum available grant is £6,800 per hectare. Proposals for planting are scored against local priority targets and must reach a minimum threshold score of 12 points. Flood risk management is a key objective and 4 points are awarded for planting catchment woodland on soils with a high propensity to generate rapid run-off – as identified by opportunity mapping (Broadmeadow et al. 2014); higher points are awarded for riparian and floodplain woodland (see Sections 3.4 and 3.5). Additional points (9 points) are given for partnership working, including where the land falls within a Making Space for Water demonstration project area or within a Woodland for Water Priority catchment. The scoring is area-based and takes into account the design of the woodland in terms of effectiveness for reducing flood flows (in terms of targeting run-off pathways and sediment sources). Additional payments are available as a Higher Tier Option under Countryside Stewardship, including a multi-year annual payment of £200 per hectare for up to 10 years to support the

successful establishment and maintenance of new woodland. Funding is also provided through the Higher Tier for woodland improvement and a range of capital items, including woodland infrastructure. Lastly, a capital grant can be received to support the development of a woodland management plan.

The Forestry Grant Scheme provides grants for planting trees and the sustainable management of existing woodlands in Scotland. Two categories with 9 options (depending on woodland type) deal with woodland creation designed to bring economic, environmental and social benefits, including protecting soil and water. There is an initial planting payment and an annual maintenance payment for 5 years, as well as a range of capital grants for operations such as fencing and tree protection. Higher payment rates are available for planting within target areas and range from £3,240 per hectare for productive broadleaves, £2,160 per hectare for conifers and £630 per hectare for low density native broadleaves (not including capital payments for items of infrastructure, such as fencing and gates). Target areas include those identified by Forestry Commission Scotland/SEPA opportunity mapping projects as where woodland creation is likely to provide multiple benefits for NFM and water quality. Reduced initial planting and annual maintenance payments apply for conifer and some native woodland creation schemes that exceed 300ha in size to reflect economies of scale and value for money. Scoring criteria (including scale of delivery, contribution to Scottish Government objectives, specified benefits and nature of work practices) are used to assess schemes and ensure the most cost-effective use of available funds.

In Wales, funding for catchment woodland is provided by the Welsh Government under the Glastir scheme, including Glastir Woodland Creation, Glastir Woodland Management and Glastir Woodland Restoration. A woodland creation opportunity map underpins a scoring system to identify preferred areas, excluding sensitivities and constraints to planting. Applications are subject to an Expression of Interest, which is separately scored according to how the proposed planting delivers against specific objectives. This allows schemes to be ranked for consideration against available funding. Financial support is provided for new planting, fencing, annual maintenance for 12 years and premium payments. Successful applicants are required to prepare a Woodland Creation Plan through which funding can be claimed for eligible items. Payment rates for woodland (£3,000–4,500 per hectare) and agroforestry (£1,600 per hectare), native woodland (£3,000–4,500 per hectare) and agroforestry (£1,600 per hectare). Land that qualifies for compensation for income forgone from stock exclusion can receive an annual premium payment of £350 per hectare for 12 years.

A number of other funding sources support catchment woodland planting aside from the RDP. One of the main sources is the UK Woodland Carbon Code, which promotes carbon trading (Forestry Commission 2014). Trees planted under the RDP may be eligible as a woodland carbon project, providing the carbon funding sought is necessary additional funding for woodland creation. This 'additionality test' ensures that there is not an issue of double funding.

Another source is the Woodland Trust, which provides advice, finance and practical support for woodland creation. The Trust operates a MOREwoods scheme that is designed to support the targeted planting of smaller areas of trees and woods of at least half a hectare. The scheme covers 50–60% of the planting cost, with the lower figure applying to contracted work that includes 2 years of maintenance work. Funding is limited to native broadleaves.

The Environment Agency has funded forestry agents in north-east England to promote woodland creation for reducing flood risk and diffuse pollution pressures in priority areas. There is also scope for supplementary funding in the form of locational premiums to be provided from regional flood levies or direct payments from the national flood defence budget, although these have not contributed to date. A key issue is State Aid Rules, which place a general limit of 80% on the proportion of woodland planting costs that can be

publically funded. The RDP aims to cover at least 50% of the actual costs involved but the schemes offering enhanced rates already approach the 80% limit, leaving little room for additional payments from other sources. Unfortunately, State Aid Rules do not recognise the additional public benefits provided by woodland creation, such as for flood risk management, which limits the scope for compensating landowners for the losses incurred in land use change.

One way of capturing the costs and translating these into appropriate incentives would be to develop a system of payments for NFM and wider ecosystem services. Forest Europe (MCPFE 2007) recommends the development, testing and implementation of such schemes to broaden and diversify the financial basis for maintaining and promoting the protective functions of forests. However, progress is constrained by the difficulty of calculating and placing a value on the different services provided. There is an urgent need for more research on quantifying woodland services and designing workable schemes. Ideally, these should integrate the full range of woodland benefits and incorporate any costs. A new EU Cost Action 'PESFOR-W' is addressing this subject, although with a focus on water quality rather than flood risk management.

The RDP does not provide grants for the planting of short rotation (<8 years) biomass crops or to grow Christmas trees.

3.2.14 Design, management and maintenance

Funding schemes such as the RDP place conditions on the design, maintenance and management of woodland creation which, if not met, will result in grants being claimed back. Conditions usually cover minimum planting areas and stocking densities, amounts of open space and shrubs, and diversity of tree species. All schemes and thereby woodland practices employed must comply with the UK Forestry Standard (UKFS) and its underpinning set of guidelines, including on water and soil protection (Forestry Commission 2011). These regulations cover the planning, design and sustainable management of woodlands based on internationally recognised science and best practice. The UKFS also underpins the independent UK Woodland Assurance Standard, which is used for voluntary independent certification. Those in receipt of grant payments must have management control over the land involved for the duration of the contract and the full commitment period. Planted trees are subject to inspection and expected to be maintained to allow proper establishment, including reasonable height growth. Maintenance payments require weeding, protection from damage such as by grazing/browsing, regular inspection, replacement of dead trees, and a diary kept of work done.

Once woodland is established, a felling licence is usually required when the landowner eventually wishes to harvest the trees. It is an offence to fell trees without a licence unless a specific exemption applies. The licence will normally include a condition to replant the area with trees and to maintain these for a period not exceeding 10 years. Applications to fell trees in order to convert land for a change in use are subject to the Environmental Impact Assessment (EIA) Regulations. If the conditions of a felling licence are not complied with, the forestry regulator may issue an enforcement notice; failure to comply with this can result in a significant fine.

3.2.15 Other requirements

Woodland creation is subject to a range of constraints and sensitivities that influence the nature of planting and thereby its scope to contribute to flood risk management. Constraints affect locations where the creation of sizeable areas of woodland is either not possible or very unlikely due to existing land use, land ownership or the presence of vulnerable assets. These include features such as the built environment, transport infrastructure, pipelines and

cables, open water, deep peat soils, Scheduled Ancient Monuments and existing woodland. Other areas may be subject to sensitivities, which affect the scale, type and design of planting. Important examples are:

- the most valuable agricultural land (for example, Grade 1)
- sites close to flood defence and urban infrastructure, which require access and may be vulnerable to the backing up of floodwaters
- areas scheduled or recognised for their nature conservation, historic or cultural importance

These aspects require careful consideration on an individual site basis in consultation with relevant agencies; this forms part of the normal assessment and approval process for woodland planting applications. Opportunity mapping can be used to help identify priority areas for woodland creation to reduce flood risk and provide other benefits (Broadmeadow et al. 2014).

Proposals for new planting, deforestation and the construction of forest roads and quarries come under the forestry provisions of the EIA Regulations. Forestry proposals that may have significant environmental impacts require an EIA before approval is granted. Regardless of the need for an EIA, anyone can comment on a woodland proposal before a decision is reached. The minimum consultation requirement in Great Britain is that grant applications, clearfelling proposals and forest management plans are entered onto a Public Register of New Planting and Felling. In addition to the Public Register, local authorities and other statutory bodies are sent details of proposals under formal consultation and notification procedures to provide an opportunity to express their views. The majority of applications, which are often subject to amendments, are approved through this process. If objections are lodged and sustained, the Forestry Commission may ask for advice from an advisory committee and/or refer to the appropriate forestry minister before arriving at a decision.

Most woodland planting, natural regeneration and some management operations that are not part of the public forest estate receive funding under the RDP. The payment of grants is conditional on meeting UKFS requirements, which include legal and good forestry practice requirements for each of 7 elements of sustainable forest management, categorised as biodiversity, climate change, historic environment, landscape, people, soil and water (Forestry Commission 2011). These elements are covered by separate sets of guidelines, which provide more details on how woodland owners, managers and practitioners can comply with the requirements.

As catchment size increases, achieving a sufficient level of woodland creation to deliver a significant reduction in downstream flood risk will require a longer term strategy and catchment plan. Co-ordinated planting by a number of landowners over time at a landscape scale would be needed as part of an integrated approach involving other NFM and traditional engineering measures, which may necessitate a higher level of grant support than is currently offered.

3.2.16 Research gaps

The most important gaps and needs in research on catchment woodland are set out below.

- Establish a research quality, long-term catchment study to measure the effects of strategic and targeted woodland creation at the small to medium catchment scale on flood flows.
- Explore technological developments in sensors and the use of remote sensing to improve measurements of flood flows during more extreme events, as well as

the assessment of woodland effects on flood generation and conveyance processes.

- Improve the way that hydrology, hydraulic and coupled models represent woodland hydrological processes in terms of process inclusion (water use/evaporation, soil infiltration, surface roughness and ideally sediment interactions) and the selection of appropriate parameter values. Critically evaluate and test the upscaling of these processes to the catchment level.
- Draw on and supplement hydrological process studies to establish and check woodland process relationships to aid modelling and the prediction of woodland effects on flood flows.
- Extend efforts at exploring and quantifying the opportunities and risks of woodland creation and other NFM measures in desynchronising or synchronising subcatchment flood flows.
- Apply existing and improved coupled models to build on recent efforts to estimate the costs and benefits of woodland creation for reducing flood risk, including the provision of other ecosystem services. Use these models to integrate the effects of different NFM measures with more traditional engineered approaches to facilitate option testing and design optimisation for flood risk management.
- Continue established research catchment and monitoring studies to measure the longer term effects of woodland creation and management on hydrological processes and flood flows.
- Better integrate NFM and water quality/diffuse pollution studies for more effective measurement of multiple water benefits.

3.3 Cross-slope woodland

3.3.1 Understanding the science

Catchment studies

The same woodland processes whereby catchment woodland can impact on flood flows also apply to cross-slope woodland, albeit on a much smaller scale. Woodland water use is likely to be enhanced per unit area by the larger woodland perimeter and thus edge effect (Nisbet 2005), as may the effects on soil infiltration and sediment retention, depending on alignment and width of the woodland in relation to surface run-off pathways. The contribution of hydraulic roughness can be expected to be similarly dependent on the structural characteristics of the individual woodland.

The localised nature of cross-slope woodland makes it extremely difficult to measure its impact at the catchment scale. This is reflected in an apparent absence of measured data quantifying the impact of this form of woodland on catchment flood flows. Only one study was found that addressed this subject at a process level and then used modelling to upscale and predict the effects on catchment flood flows. This was the Pontbren study in mid Wales. Marshall et al. (2014) found soil infiltration rates to be 67 times higher within woodland plots and shelterbelts planted on improved grassland than on grazed pasture, which reduced measured run-off volumes by an average of 78% compared with the control. These differences were quick to develop, becoming apparent within one year of sheep exclusion

and tree planting. This was partly explained by the removal of the grazing pressure on the soil, which reduced run-off volumes by 48%, and partly by the action of tree rooting and growth, which was responsible for the remaining 30% decrease. Observations indicated that the development of the vegetation in the ungrazed and tree planted plots led to increased times to peak. Soil measurements in the tree planted shelterbelt revealed significantly higher hydraulic conductivity values (x2.4) compared with the grazed pasture. This was associated with a greater proportion of larger soil pores and flow pathways provided by the tree roots (Solloway 2012).

Modelling studies

The measured data from the hillslope plot studies at Pontbren were used to develop and parameterise a physics-based, distributed run-off generating model which incorporated water use, soil infiltration and surface roughness processes. The model simulated the effect of planting tree strips across 7% of the 12km² headwater catchment on a severe flood in the form of the January 2005 event at Carlisle in north-west England (140mm of rainfall over 2 days, with an estimated return period of 180 years). Although this event was not observed at Pontbren, the model predicted that the tree strips would reduce the flood peak by an average of 5% (with 95% confidence intervals of between a 2% and an 11% reduction) at the headwater scale (McIntyre et al. 2012).

Other modelling studies have simulated the effect of targeted woodland planting within catchments on peak flows but not involving cross-slope woodland. For example, an updated version of the MIKE SHE/MIKE11 model was used to assess the effect of tree planting on steep slopes within the River Tone catchment, a tributary of the River Parrett in south-west England (Park et al. 2009, McIntyre and Thorne 2013). As at Pontbren, water use, infiltration and surface roughness processes were incorporated into the model, although no information is provided on the selected parameter values. Tree planting on 19–37% of the catchment was predicted to have little effect on the largest peak flow event in January 2002 (varying from a 1% increase to a 2% reduction in flow), but reduced the largest event in May 2002 by between 5% and 21%. The difference in results was ascribed to seasonal effects, particularly the higher woodland interception and transpiration rates in summer periods which led to lower pre-event/antecedent soil wetness.

Another example was the application of a physics-based, run-off generating model to the Hodder catchment in northern England. This simulated the effect of woodland planting on mineral soils occupying 29% of a 25km² subcatchment on the average of the 10 largest peak flows within a one-year period. Changes in land cover were represented by altering model parameters for soil infiltration rate, canopy storage and surface roughness, with values drawn from the literature (Ballard 2011). The model predicted that planting conifer woodland on these soils would reduce peak flows by an average of 7% (with 95% confidence intervals of between a 3% and a 13% reduction) compared with a 4% reduction with planting broadleaved woodland (with 95% confidence intervals of between 0% and a 9% reduction).

3.3.2 Confidence in the science

There is high confidence in the process understanding and the measured data from the hillslope plot studies at Pontbren that cross-slope woodland can reduce surface run-off. This provides medium confidence in the ability of this type of woodland to reduce small to moderate flood flows in small catchments (<10km²). Modelling studies incorporating process knowledge and data suggest that cross-slope and other targeted forms of woodland planting could also reduce large flood peaks (perhaps as high as a 0.5% AEP flood) in small catchments, although the level of confidence is relatively low due to the uncertainties in upscaling process knowledge. Local stakeholders tend to display a higher level of

confidence in the role of cross-slope woodland based on physical observations by the farming community at Pontbren of the contrasting response of surface run-off from sheep pastures compared with tree shelterbelts during high rainfall events. The size of the reduction in peak flows achievable will depend on the extent of cross-slope woodland in a catchment.

The level of confidence in this more spatially restricted form of woodland declines sharply with increasing catchment size. Modelling suggests reductions in winter flood peaks are likely to be <10% for targeted woodland planting involving <30% catchment cover, while summer floods could perhaps be reduced by up to 20%. These magnitudes of reduction, especially for winter events, lie within the margin of measurement error and are thus very difficult to validate. Nevertheless, it represents a 'real' effect which could contribute to flood risk management.

A related issue is that the size of flood reduction achievable with woodland creation will depend on the nature of the existing land use and standard of management practices employed. The more extensive the soil damage in terms of soil compaction and sealing, the greater is the likely reduction in peak flows following planting. This factor will increase the margin of uncertainty, especially in larger catchments.

Overall, there is likely to be low confidence in the ability of cross-slope and other targeted forms of woodland creation to reduce large flood peaks in large catchments.

3.3.3 Metrics of cost–benefit and multiple benefits

No information could be found on cost-benefit metrics for cross-slope woodland at Pontbren or elsewhere. This type of woodland could be expected to deliver a range of other environmental and related benefits as described for catchment woodland in Section 3.2, although at a reduced spatial scale. For example, cross-slope woodland can be very effective in removing sediment and other diffuse pollutants in surface or near surface run-off from upslope land, thereby improving water quality and potentially contributing to reduced flood risk through reduced sediment delivery to watercourses (Nisbet et al. 2011b). It can also be useful for providing shelter for livestock and crops, depending on orientation and predominant wind direction, as well as forming a useful conduit and refuge for wildlife, linking isolated blocks of woodland habitat to create a habitat network.

3.3.4 Effectiveness/performance

While cross-slope woodland as a component of catchment woodland will contribute to peak flow reduction on the basis of its relative woodland footprint, there is potential for its effectiveness to be greater on a unit area basis by virtue of its targeted placement in the landscape. This will depend on the same set of factors outlined in Section 3.2 for catchment woodland; more specifically, it will be strongly influenced by certain aspects of woodland design, most notably the width of woodland belt/strip, its relative orientation/alignment across the slope, and placement with respect to run-off sources and pathways. Woodland type, structure and density of planting will also be key factors, as will the degree of slope and volume of upslope run-off. All of these will interact, with the effectiveness of cross-slope woodland for reducing hillslope run-off and peak flows likely increasing with:

- width of woodland belt perpendicular to dominant surface or near surface run-off paths
- multiple woodland belts spaced appropriately downslope in relation to volume of run-off

- the capacity of the soil in terms of depth, structure and texture to receive and store run-off volumes
- the degree of baseline soil damage and thus the propensity of the site to generate rapid run-off

For the reasons noted earlier, measurements of the contribution of cross-slope woodland to flood risk management are best made at the plot/hillslope level and then upscaled using appropriately parametrised and calibrated run-off generating models.

3.3.5 Key case studies

Only one key case study could be found. This was the **Pontbren Case Study in mid Wales**. This has been selected for the <u>Evidence Directory</u>, where a detailed description of the nature of the study can be found. A new study has been established at Tebay in Cumbria, though this involves low density woodland planting on steep slope sections rather than cross-slope woodland belts. No flow measurements are involved but there are some site assessments of soil hydrology, which will aid future modelling.

3.3.6 Funding

Funding for cross-slope woodland through the RDP is potentially constrained by woodland area and width limits set by individual countries. Countryside Stewardship in England has a minimum block size of 0.1ha, minimum width of 10m and minimum area per application of 1ha. The Forestry Grant Scheme in Scotland for 'Small or Farm Woodland' has a minimum block size of 0.25ha and minimum width of 15m; while Glastir in Wales (including an 'Agroforestry category') has a minimum block size of 0.1ha, minimum width. Other limits apply to woodland type and stocking densities, while lower rates of grant, restrictions on fencing payments and lower scores for smaller applications may apply in some countries.

The Woodland Trust's MOREwoods scheme provides financial support for planting small areas of trees and woods, although the minimum qualifying area is 0.5ha and is restricted to the planting of native broadleaved trees.

The UK Woodland Carbon Code launched a Small Woods Scheme Pilot in September 2015.¹⁵ This has a minimum block size of 0.1ha, minimum width of 20m and maximum block size of 5.0ha. However, smaller projects can be grouped up to a maximum area of 50ha, with a minimum stocking density of 1,200 trees per hectare.

The other potential funding opportunities listed in Section 3.2.13 are less likely to be applicable due to the small size and distributed nature of cross-slope woodland.

3.3.7 Design, management and maintenance

As described in Section 3.2.14 for catchment woodland, these requirements are covered by the conditions applying to the different grant schemes that fund woodland creation, including abiding by the UKFS. While the requirements are primarily focused on achieving successful woodland establishment for multiple benefits, in some cases they are tailored to increasing effectiveness for reducing flood risk and diffuse pollution. For example, the design principles for planting woodland for water under Countryside Stewardship in England advises on higher planting densities along surface run-off pathways to increase hydraulic roughness, as well as incorporating an edge strip of grassland to enhance the trapping of fine sediment in overland

¹⁵ <u>https://www.forestry.gov.uk/forestry/beeh-a26gjb</u>

flow. Depending on size/width, cross-slope woodlands are more likely to be managed as semi-permanent features subject to minimum intervention, preserving their shelter, water, habitat and other benefits. Opportunity mapping has an important role to play in identifying priority locations for planting (Broadmeadow et al. 2014).

3.3.8 Other requirements

The smaller scale and potentially more integrated nature of cross-slope woodland means that it is less likely to be affected by some land use constraints and most sensitivities (such as landscape and conservation designations). It is also more likely that planting applications will fall below the threshold for an EIA. As a result, the process and therefore the timescale for approving planting schemes may be much quicker, aiding the development and potentially the upscaling through multiple schemes of the woodland contribution to reducing flood risk.

3.3.9 Research gaps

The most important gaps and needs in research on cross-slope woodland are set out below.

- Undertake additional hillslope-scale studies to measure the effects of planting cross-slope woodland in different settings (for example, in terms of slope, position on slope and farming practice) on run-off/flood generation to check the transferability of results from Pontrren.
- Supplement the above hillslope-scale studies with assessments of the effects of woodland design and management on run-off/flood generation, particularly the width of cross-slope woodland, woodland type and changes with woodland age.
- Use appropriately parameterised and calibrated hydrology models to explore the effects of woodland design and management factors, as well as woodland placement, on run-off/flood generation. Incorporate how cross-slope woodland affects water use and surface roughness processes within these models and integrate effects.
- Establish a research quality, long-term catchment study to measure the effects of the planting of a series of cross-slope woodlands within a small catchment on peak/flood flows. Such a study could be nested within a larger scale study of the impact of catchment woodland on flood flows (see Section 3.2.12).
- Use the data from this catchment-scale study to critically evaluate and test the ability of hydrology models to upscale process understanding and knowledge from the plot/site level to the catchment level to predict the effects of cross-slope woodland on a range of flood flows.

3.4 Riparian woodland

3.4.1 Understanding the science

Catchment studies

The water benefits of riparian woodland are well known (Broadmeadow and Nisbet 2004). They include:

- protection of water quality
- sediment removal and erosion control
- moderation of shade and water temperature
- maintenance of habitat structural diversity and ecological integrity(aids the retention and delays the passage of floodwaters downstream)

These benefits have been well studied at the reach level but much less researched at the catchment scale, largely reflecting the difficulty of separating the riparian signature from the larger catchment area. No BACI type catchment studies could be found that have measured the effect of riparian woodland planting or removal on flood flows.

Some small catchment studies aimed at quantifying the impact of planting on flood flows have been conducted in the UK in recent years, most notably at Pickering in north Yorkshire and Eddleston Water in the Scottish Borders. The planted trees in these cases are too young to be expected to generate any significant effect and any detailed analysis awaits the collection of longer term data. The ability to detect an eventual effect is likely to be constrained by a number of factors, including:

- the absence of flow measuring structures (increasing measurement errors)
- difficulties in separating out the impact of other potential land use and management changes in the wider catchment
- the lack of a proper control catchment to allow for background changes in climate and so on

A number of other NFM schemes involving the creation or restoration of riparian woodland are underway or planned but include only qualitative data or no monitoring (JBA Consulting 2015).

Reach level studies

At the reach level, the focus of studies concerning interactions with flood flows has been on the specific contribution of LWD, especially where this forms dams. LWD is an integral part of naturally functioning riparian woodlands, although the tradition in the UK has been to actively remove it from river channels where there are concerns about interference with fisheries or the risk of washout and downstream blockage of pinch points. LWD is known to exert a significant effect on channel flows and processes, as it affects channel development, sediment deposition, bank scour and overbank flows. See Chapter 4 for an assessment of the evidence on the impact of LWD on flood risk management.

Separate process studies have examined the water use of riparian trees, how they influence water velocity and water levels in the riparian zone, and the deposition of sediment. Brown (2013) showed how riparian trees can maintain high evaporation losses, creating potential additional below-ground water storage, especially in summer periods. Above-ground water storage is increased by the friction/drag of riparian trees, which slows water flows and increases water levels, although this can be partly offset by enhanced channel velocities, depending on the presence of LWD dams (Thomas and Nisbet 2006). By slowing water flows, riparian woodland is also effective at enhancing sediment deposition on the floodplain, reducing downstream siltation within river channels (Piégay and Bravard 1996). Soil infiltration rates during winter periods are often constrained by the high water table within the riparian zone, limiting the contribution of this potential benefit of riparian woodland.

Modelling studies

Modelled data provide the best source of evidence that riparian woodland can reduce flood flows at the catchment scale. Odoni and Lane (2010) predicted that the NFM measures of planting 50ha (0.7% of the catchment) of riparian woodland (with an assumed natural density of LWD dams) plus the construction of 100 LWD dams within existing stretches of riparian woodland in the Pickering Beck catchment in north Yorkshire would reduce a 4% AEP flood by 4% and a 1% AEP event by 8%. The larger reduction for the more extreme event was explained by the greater interaction between flood flows and riparian trees as a result of more water coming out of bank. There was a synergistic effect between the action of the riparian trees and the associated LWD dams, with the latter making a relatively greater contribution to the overall impact.

Other modelling studies include those reported by McIntyre and Thorne (2013) in their review of land use management effects on flood flows and sediments. The application of an enhanced version of the MIKE SHE/MIKE 11 model in the River Tome catchment in Somerset predicted that tree planting on riparian areas occupying between 5% and 9% of 4 subcatchments would surprisingly increase the size of the largest peak flow event in January by between 2% and 3%, but reduce the largest event in May by 1% to 2%. The reason for the predicted increase in the January peak flow is unclear, but the authors noted the smaller woodland water use and wetter soils in the winter period, as well as the inherent uncertainties in the model application and optimisation process (that is, the small predicted positive and negative changes fell within modelling errors). A separate study using a different physics-based model predicted that the planting of deciduous riparian woodland on 9% of the 25 Hodder catchment in northern England would reduce the average of the 10 largest peak flows within a one-year simulation by a mean of 2% (with 95% confidence limits of between a 0% and a 3% reduction).

Dixon et al. (2016) applied a spatially distributed flood model (OVERFLOW) to analyse the effects of river restoration and woodland creation on flood flows within the 98km² catchment of the Lymington River in southern England. They found that the restoration of riparian woodland along 20–40% of the total catchment area was the most effective of the NFM measures tested, reducing peak flows by up to 19% for a 3% AEP flood. In other studies, Ghavasieh et al. (2006) showed that riparian woodland strips along a 20 reach could reduce peak discharge by 3.8%, while Anderson et al. (2006) predicted that tall vegetation reduced peak discharge by 12% over a 50km reach.

Model predictions are supported by site observations and survey-based assessments of the interactions between riparian woodland and recorded flood events. Piégay and Bravard (1996) examined the response of a Mediterranean riparian forest in France to a 0.25% AEP flood and found that the forest reduced the morphological and flow capacity for this large flood by enhancing sediment deposition, increasing hydraulic roughness and spreading flows. While the size of event caused the washout and loss of the riparian woodland within a core active tract, the woodland was expected to be more stable and effective during lower magnitude floods.

Modelling has demonstrated that the relative placement of riparian woodland within a catchment has a significant influence on the magnitude of the effect on peak flows by synchronising or desynchronising subcatchment flow responses. Odoni and Lane (2010) found that riparian planting within the lower reaches of the Pickering Beck catchment would increase peak flows by delaying the evacuation of waters to bring them into phase with those draining the upper catchment. Planting in the upper catchment delivered the greatest benefit, by spreading out the flow response and thus lowering the overall flood peak. Targeting the middle reaches produced a neutral or beneficial effect. Similar findings were obtained by Dixon et al. (2016), who observed that the largest reductions in peak flows resulted from

placements designed to maximise the desynchronisation of the timings of subcatchment flood waves, which typically involved areas in the middle and upper catchment.

It is thought likely that modelling studies underestimate the impact of riparian woodland on flood flows by not incorporating the full range of woodland processes. For example, some models like OVERFLOW consider only changes to hydraulic roughness, while none include sediment interactions, which can be particularly significant for this form of woodland. There is also uncertainty about the values selected by modellers to represent different woodland parameters as these are rarely reported in published papers.

3.4.2 Confidence in the science

There is medium confidence based on observations at the reach scale plus modelling studies that riparian woodland has the potential to reduce small to medium flood flows in small and medium sized catchments. The magnitude of effect, however, may be relatively small (<5%), unless the woodland is appropriately placed to maximise the desynchronisation of subcatchment contributions. Uncertainty about model parameterisation and the approach to evaluating flow timing effects, as well as concern over the scope for negative impacts, results in low confidence among water regulators. Some regulators view riparian woodland as more likely to increase rather than reduce flood risk due to the threat posed by the washout of LWD. Levels of confidence are more variable among other stakeholders, with some local communities – such as at Pickering – believing that riparian woodland can make a significant contribution to reducing flood peaks, while those affected by flooding associated with the blockage of structures by LWD hold the opposite view.

In general, there is low confidence that riparian woodland can exert a significant effect on flood risk at the large catchment scale. This mainly reflects the reduced footprint and interaction between the riparian zone and river flows as channel width increases, especially along Main Rivers protected by river embankments and other flood defences. In fact, there are legal restrictions on planting trees close to Main Rivers due to the need to maintain embankments for flood protection and channel conveyance.

Riparian woodland is considered unlikely to reduce large and especially extreme floods, probably regardless of catchment size. This is partly due to its relatively small extent/area but also to the increased risk of tree and LWD washout during very high flows.

3.4.3 Metrics of cost–benefit and multiple benefits

No data on the metrics of cost-benefit for riparian woodland could be found apart from those generated by the Slowing the Flow at Pickering study, which are only available for the combined set of woodland measures (see Section 3.2.10). The multiple benefits of riparian woodland for both the water and wider environment are well known (Parrott and MacKenzie 2000, Broadmeadow and Nisbet 2004), but have yet to be specifically quantified in terms of economic value.

3.4.4 Effectiveness/performance

The effectiveness of riparian woodland at reducing flood flows depends on many of the factors already described for catchment and cross-slope woodland. The most important factors are length of riparian woodland, width, structure, tree spacing, species mix, amounts of deadwood and management regime.

Woodland type is not an issue as the UKFS favour the establishment of native riparian woodland, which is predominantly broadleaved in character. While this limits the water use

effect, there is scope to enhance evaporation losses and soil drying by favouring waterdemanding species such as willow and poplar where water resources are not an issue.

The length of riparian woodland determines the overall size of footprint, which will be reduced where the woodland is limited to one bank.

Width, structure and tree spacing mainly have an impact on hydraulic roughness and sediment retention, and act as per cross-slope woodland. The relative depth of roughness elements is more important for riparian woodland for interacting with overbank flood flows, and can be increased by the presence of multi-stemmed trees, low branches and LWD on the floodplain. Woodland age and management thus have a key role to play by influencing the supply and retention of deadwood, as well as the growth and nature of the tree canopy, understorey and ground vegetation. The formation and management of LWD dams are also very important (see Chapter 4).

Important catchment factors affecting the performance of riparian woodland are:

- relative location (with respect to potential synchronisation and channel size see above)
- slope
- channel gradient
- floodplain width

The last 3 of these influence the speed, volume and space for floodwater, as well as the degree of connection with riparian woodland. Lower gradients and wider floodplains tend to enhance the interaction between the woodland and flood flows, slowing response times and increasing flood storage. Such conditions are also more conducive to the formation and action of LWD dams.

A number of small and medium sized catchment studies are underway to try and quantify the effect of riparian woodland planting on flood flows (see Section 4.5 and Case Study documents). As noted in Section 4.1, it is very difficult to separate the effects of this measure from other changes, especially given the expected magnitude of the contribution relative to the size of measurement errors involved, as well as the length of time for the woodland to grow and become fully effective.

3.4.5 Key case studies

The most complete case study of the effects of riparian woodland on flood risk in terms of observations, measurements, modelling and cost benefit is the **Slowing the Flow at Pickering** project in north Yorkshire. A detailed description of the study can be found in the
<u>Evidence Directory</u>, along with information on a second key study that is part of the **Eddleston Water Project** in the Scottish Borders.

A number of other NFM studies include an element of riparian woodland, but either its contribution is not separately determined or the planting is too recent/young to display any effects. Riparian woodland creation schemes are increasingly being established to provide multiple water and wider benefits, but seldom justify detailed flow monitoring due to the relative size of planting involved and the long timescale for effects to fully develop.

3.4.6 Funding

There is no special funding for riparian woodland creation or management within country RDPs or other schemes, which are subject to the same payment rates as those given in

Section 3.2.13. As noted for cross-slope woodland in Section 3.3.6, planting riparian woodland is potentially constrained by the minimum widths and areas that apply to these grant schemes, as well as possibly by restrictions on fencing payments. The linear nature of riparian woodland can significantly increase fencing costs, while maintenance costs can be higher due to tree losses during early flood events, as well as flood damage to fencing. Country scoring under RDP can both support and discourage riparian planting depending on national priorities and scheme size.

Separate funding for planting riparian woodland can be provided by water regulators as part of river restoration schemes, aimed primarily at meeting EU Water Framework Directive objectives.

3.4.7 Design, management and maintenance

As noted in Sections 3.2.14 and 3.3.7, these aspects are covered by the conditions applying to the different grant schemes, as well as by UKFS regulations and guidelines. Riparian woodland will often be designed and managed as semi-natural woodland habitat and a riparian buffer, with low intervention. An initial high density of planting may be favoured within floodplain areas to enhance hydraulic roughness, with subsequent thinning to support structural development and to control shade levels. Where possible, watercourses should be left to migrate and develop side channels and other natural features that will help to improve connectivity with the woodland. This will further delay and increase the storage of floodwaters, provided it is acceptable to landowner(s) and does not present a local flood risk for adjacent properties.

Intervention may be required to construct and maintain associated LWD dams, depending on the degree to which these are left to naturally develop and move. A greater degree of management will be required where these features are specifically designed and thus need to be maintained for reducing downstream flood risk, or where the washout of debris poses a particular threat of blocking downstream structures. This work may be grant aided under future RDPs (see Chapter 4 for more details).

3.4.8 Other requirements

The broadleaved nature and multiple water and wider benefits provided by riparian woodland mean that planting applications are likely to face fewer issues and objections. The main problem is likely to be the potential for riparian woodland to have an adverse impact on flood risk management, including by:

- · restricting access for maintenance of river embankments or other flood defences
- reducing channel conveyance where this can enhance local flood risk
- the backing up of floodwaters where this can flood upstream assets
- the washout of woody debris blocking downstream structures
- the synchronisation of subcatchment flows increasing downstream flood peaks

These issues can potentially be controlled through care over placement and management, aspects of which are already covered by the need for consent, or by water regulator authority guidelines. A remaining source of uncertainty is the stability of LWD dams and the relative mobility of released LWD, as well as the effectiveness of different structures for trapping and retaining loose debris.

Other issues affecting riparian woodland creation include the potential loss of existing open wetland habitat and the desire to maintain valued open landscapes.

3.4.9 Research gaps

The most important gaps and needs in research on riparian woodland are set out below.

- Establish a research quality, long-term catchment study to measure the effects of planting an extended/complete length of riparian woodland within a small catchment on peak/flood flows. Such a study could be nested within a larger scale study of the impact of catchment woodland on flood flows (see Section 3.2.16).
- Either separately or as part of this catchment study, establish site studies to measure the effects of native riparian woodland on a number of hydrological processes, including water use/evaporation, soil infiltration, soil water storage, soil erosion/sediment delivery and hydraulic roughness (including interactions with LWD dams).
- Improve the way that coupled hydrology and hydraulic models represent riparian woodland processes in terms of both process inclusion (water use/evaporation, soil infiltration, surface roughness and ideally sediment interactions) and the selection of appropriate parameter values. Use data from the long-term catchment study to calibrate and test models, as well as to explore the effects of riparian woodland length and placement on flood flows.
- Use appropriately parameterised and calibrated models from the site studies to explore the effects of varying riparian woodland design (for example, adjusting mosaic of closed canopy woodland, scrub thicket, trees with open glades and open ground) and management (for example, adjusting amounts of deadwood, as well as the structure/design and numbers of LWD dams) on flood flows.

3.5 Floodplain woodland

3.5.1 Understanding the science

Catchment studies

Floodplain woodland influences flood flows in a similar way to riparian woodland, but with a larger footprint. Floodplain processes dominate over river channel interactions arising from the greater contact between flood flows and the woodland. The tendency for floodplain width to increase down river systems means that the scope for floodplain woodland to have an impact on flood flows is usually greatest in middle and lower river reaches, and thus for medium to large catchments. However, this inevitably makes it more difficult to measure the effect of floodplain woodland on catchment flood peaks, particularly given its limited extent in the UK and the disconnected nature of much of lowland floodplains due to river embankments and other flood defences. More extensive areas of floodplain woodland are found on mainland Europe and elsewhere in the world, but are mainly associated with very large and controlled river systems which similarly make catchment-scale measurements very difficult.

Reach level studies

No measured data appear to be available at the catchment scale quantifying the impact of floodplain woodland on flood peaks. Instead measurements have focused on interactions between woodland, water flows and sediment at the reach level, or on modelling these

processes and upscaling the effects to the catchment level. Much is known about how floodplain woodland affects both floodplain and channel hydraulic roughness by the physical presence of the trees, undergrowth and deadwood, as well as by the influence of these on diverting floodplain flows and driving the formation of multiple channels and backwater pools (Piégay and Bravard 1996). Standard engineering tables show how dense, multi-stemmed woodland typical of natural floodplain woodland exerts the greatest hydraulic roughness of all vegetation types, with values of Manning's n 5 times or more greater than those for grassland (Chow 1959).

While Manning's n provides a basic measure of hydraulic roughness, the fact that it lumps together and simplifies a number of energy losses/resistance effects reduces its usefulness for looking at the role of woodland design and management factors such as tree spacing, distribution, structure (for example, branching), age and amounts of dead wood. This has led to the development of more sensitive approaches based on drag force and fluid dynamics, supported by 2D and 3D hydraulic models (Nepf 1999, Rameshwaran and Shiono 2007, Wilson et al. 2008). Laboratory-based flume and process modelling studies by Xavier et al. (2007) and Whittaker et al. (2013) demonstrate how the size, placement and orientation of floodplain trees affects energy loss by resistance and turbulence, reducing water velocity and raising local water levels on the floodplain. This work is important for optimising woodland design for reducing flood flows.

Most modelling studies have relied on Manning's n to represent hydraulic roughness and assess the impact of floodplain woodland on flood flows. Thomas and Nisbet (2006) used a 2D hydraulic model to simulate the effect of the increased hydraulic roughness associated with planting native floodplain woodland along a 2.2km grassland reach of the River Cary in Somerset on a 1% AEP flood. The woodland was predicted to reduce water velocity by 50% and raise the flood level within the woodland by up to 27cm, increasing temporary floodwater storage by 71% and delaying the downstream progression of the flood peak by 140 minutes. These results were considered significant for reducing downstream flood risk by potentially desynchronising flood flows and providing more time for issuing flood warnings. O'Connell (2008) also applied the River 2D hydraulic model to 3 floodplain sites in the Mawddach catchment in mid Wales and found woodland planting would increase water depths on the floodplain by between 0.5m and 1.2m, as well as delay peak discharge by >30 minutes. Another modelling study by Nisbet and Thomas (2008) at Ripon in north Yorkshire predicted that planting floodplain woodland at 4 sites in the River Laver catchment totalling 40ha in area (<1% of catchment) could delay the progression of the 1% AEP flood by around 1 hour. This had the potential to reduce the flood peak at Ripon by 1-2% by desynchronising the flood contribution from the adjacent tributary, the River Skell. It was hypothesised that a much greater reduction could be achieved by planting a larger area of floodplain woodland along the river system.

Other modelling studies have assessed the impact of planting larger areas of floodplain woodland and found this to have a small or no effect on large flood peaks. For example, work by Park and Cluckie (2006) in the River Parrett catchment in southwest England predicted that converting a 200m wide zone of existing floodplain grassland or arable to woodland would have no effect on flood risk. Another study in the Laver catchment in northern England found that converting 25% of the floodplain to broadleaved woodland would only reduce the 1% AEP flood by 1–2% and delay the flood peak by 15 minutes (JBA Consulting 2007). Similarly, Johnson (2006) predicted that large-scale planting along the floodplain of the River Enrick catchment at Glen Urquhart in north Scotland would reduce a 0.5% AEP flood by 0.8%, while the flood peak was delayed by 1 hour. In contrast, Dixon et al. (2016) found that the restoration of floodplain woodland within subcatchments comprising over 10–15% of the area of the Lymington River catchment in southern England reduced the 3% AEP flood by 6% at 25 years post planting, with larger reductions in peak discharge occurring as the forest aged up to 100 years. They highlighted the importance of woodland

placement within larger catchments from the perspective of maximising the desynchronisation of subcatchment flood waves.

Environment Agency (2015b) modelled the effect of planting SRC willow across 3 case study floodplains on a 1% AEP flood. A linked 1D–2D hydraulic model (ISIS-TUFLOW) was used to simulate the effects of different extents and placement of coppice on the floodplain. This dense and fast-growing energy crop offers scope for rapid establishment of hydraulic roughness to interact with flood flows. It was found that a complete cover of coppice across the floodplain had the greatest impact on flood dynamics by increasing flood depth within the woodland by >20cm and reducing flood velocity by >40%. The plantation acted like a 'green leaky dam', holding back and reducing the speed of floodwater propagation. This created a backwater effect that extended up to 300m upstream of the woodland.

It is difficult to evaluate and compare the results of these studies since different models are used, and it is often unclear what value of Manning's n was selected to represent floodplain and channel roughness for the different vegetation types. Another issue is that the 2D and more often 1D representation of the floodplain in the models makes it very difficult to capture the diversity in channel structure and floodplain form typical of floodplain woodland, including the contribution of LWD. It is also notable that none of the hydraulic models include the hydrological effects of woodland, especially water use. Zell et al. (2015) showed that floodplain woodland can have a high water use, which can significantly increase the capacity for below-ground storage of floodwater, particularly during summer months. They demonstrated that a bottomland hardwood forest in the USA had 16% greater evapotranspiration and 28% more vadose zone water storage compared with an agricultural field. Sediment interactions are also seen as having an important contribution to make, with floodplain woodland being very effective at capturing/filtering and retaining river sediments, reducing downstream siltation and maintaining channel conveyance where most critical (Piégay and Bravard 1996).

There is potential for floodplain woodland to have the opposite effect of increasing flood risk by:

- synchronising subcatchment flows
- the backing of floodwaters upstream
- the washout of woody debris

As with riparian woodland, these risks can be controlled through woodland placement and possibly design.

3.5.2 Confidence in the science

There is medium confidence based on process understanding and modelled data that floodplain woodland has the potential to reduce medium to large flood flows in medium to large sized catchments. The magnitude of effect on flood flows, however, may be relatively small (<5%), unless the woodland is appropriately placed and sized to maximise the desynchronisation of subcatchment contributions. Confidence is lower about the ability to reduce small floods since this will be strongly dependent on the degree to which floodwaters come out of bank and interact with the wider floodplain woodland. The limited presence of the latter within small catchments means that it is less likely to play a role here.

In comparison, the level of confidence among stakeholders is considered to be low to medium. This reflects the expected small size of the contribution, as well as the absence of measured data at the catchment level, the large uncertainties in model applications, and the risk that planting floodplain woodland could potentially increase flooding.

3.5.3 Metrics of cost–benefit and multiple benefits

No cost-benefit metrics on the contribution of floodplain woodland to flood risk management could be found. In contrast, the multiple benefits are well known (Environment Agency 1996, Hughes 2003, Bronstert and Kundzewicz 2006), although economic valuations are largely limited to those calculated for woodland in general. These are likely to underestimate floodplain woodland benefits, which are particularly strong for biodiversity (Hughes et al. 2001) and helping to reduce diffuse pollution, including by enhancing siltation and sediment retention (Jeffries et al. 2003), nutrient (phosphate and nitrate) removal (Gilliam 1994) and fixing heavy metals (Gambrell 1994). The higher water use of floodplain woodland can pose issues for water resources, although this can be offset by the slower release of river water or increased recharge of groundwater from the greater water storage within pools, side channels and floodplain soils (McGlothlin et al. 1988).

3.5.4 Effectiveness/performance

The performance of floodplain woodland is affected by many of the factors described above for riparian and other forms of woodland. Key factors are extent and placement of floodplain woodland, as well as woodland shape, alignment and structure. In general, the greater the extent of the floodplain woodland in terms of both its width across the floodplain and its length, the greater the contribution to reducing flood flows. Restricting planting to one side of the floodplain is obviously less effective, while analyses suggest that a series of separate small blocks can have a similar effect to an equivalent single, wider block (Nisbet and Thomas 2008). Performance is likely to be enhanced where the river channel is allowed to migrate and take on a more natural form, promoting interaction and out of bank flows. Where the river remains as a single, straight channel, the flow retardation benefit from planting floodplain woodland can be partly offset by increased channel velocity.

Placement of the woodland is critical for maximising the desynchronisation of subcatchment flows, as well as for avoiding throttle points and sites where the backing up of floodwaters could affect local properties. Local variation in microtopography is also very important, with planting in the lower lying, wettest parts of the floodplain having the greatest interaction with flood flows. Floodplain woodland will have little impact on flood flows where disconnected from the river system by flood embankments or other defences. This includes locating woodland within flood storage areas such as washlands which, while marginally reducing the available storage volume, can nevertheless represent an appropriate use of the land providing the flooding regime is compatible with tree survival and establishment (Armbruster et al. 2004).

Woodland shape, alignment and structure also affect flood flow interactions, which are enhanced by a wider perimeter area perpendicular to the direction of flow, ensuring tree rows and gaps are not aligned with flows, and by an increased density of tree and shrub planting. The latter is best designed to maximise roughness for the expected range of flood depth.

For the reasons noted in Section 3.5.1, it is very difficult to measure the effects of floodplain woodland on flood flows at the catchment scale. No properly controlled before and after studies are available, with field measurements limited to a few relatively young, reach-scale, woodland planting schemes in southwest England (see below).

3.5.5 Key case studies

The relatively high agricultural value of floodplain land, restrictions on planting close to Main Rivers and a variety of other issues continue to constrain the scope for planting floodplain woodland. Consequently, there are only a couple of small-scale, case studies in place that

are attempting to measure the impact on flood flows. These are both located in the River Parrett catchment in south-west England and each involves ~5ha of woodland on one bank of the floodplain. The woodland is <10 years old, and while now established and providing significant hydraulic roughness, it remains a significant challenge to measure any changes to river flows.

In the absence of measured data for a sizeable study, the 2 model-based assessments in the **River Cary catchment in Somerset** and **River Laver in north Yorkshire** have been selected as the best case studies for demonstrating the potential benefit of floodplain woodland for flood risk management. Details of these are included in the <u>Evidence Directory</u>, along with information on a study in the Lymington River catchment draining the New Forest in southern England, which involves a number of woodland and other NFM measures.

3.5.6 Funding

The high agricultural value of drained floodplain land and high biodiversity value of unmanaged floodplain provide a significant constraint to any sizeable planting of floodplain woodland. While floodplain woodland is favoured by the scoring systems employed in some country RDPs, the absence of any special payment rate means that uptake has been low to date. Previous efforts in England at offering additional funding contributions proved unsuccessful (Nisbet et al. 2015).

There is greater scope for planting SRC within floodplain areas as this offers early and much greater financial returns than productive or non-productive floodplain woodland. Although this form of woodland planting does not receive grant support under the RDP, the growth of renewable energy may provide an attractive market to farmers for biomass for power generation. SRC provides an effective way of quickly establishing high hydraulic roughness and water use, with potential significant benefits for reducing flood flows (Calder et al. 2009, Finch et al. 2011, JBA Consulting 2015). However, an incentive may be required to help coordinate and encourage a sufficient scale of planting within priority catchments to make a difference. SRC is not eligible for payments under the UK Woodland Carbon Code.¹⁶

3.5.7 Design, management and maintenance

As noted for the other woodland measures, these requirements are covered by the conditions applying to the different grant schemes, as well as by UKFS regulations and guidelines. The wettest parts of floodplain woodland are likely to be designed and managed as native wet woodland habitat and subject to low intervention. An initial high density of planting may be favoured, followed by subsequent managed or self-thinning to encourage the formation of a diverse and multi-layered woodland structure, providing high hydraulic roughness and aiding the formation of side channels and pools. Where possible, watercourses should be left to migrate and develop side channels and other natural features that will help to improve connectivity with the woodland to further delay and increase the storage of floodwaters. A significant element of willow or poplar species would increase water use and benefit below-ground water storage. Higher elevation areas could support more productive, mainly broadleaved woodland, offering lower surface roughness but higher water use.

The planting of SRC for flood risk management would need to be designed and managed to maintain and maximise interactions with flood flows. This would include designing the layout of the coppice and the phasing of the harvesting to always retain a portion of the standing

¹⁶ https://www.forestry.gov.uk/carboncode

crop, as well as taking care to align rows and gaps so that these are not parallel to the direction of flow.

3.5.8 Other requirements

Floodplain woodland is subject to the same issues and requirements detailed in Section 3.4.8 for riparian woodland.

3.5.9 Research gaps

The most important gaps and needs in research on floodplain woodland are set out below.

- Establish a research quality, long-term, reach-scale study to measure the effects of planting a sizeable area (30–50+ hectares) of floodplain woodland, ideally across both sides of the floodplain, on flood flows. Such a study could potentially be nested within a larger scale study of the impact of catchment woodland on flood flows, although it would be better suited to the middle to lower reaches of a large catchment.
- Either separately or as part of this reach-scale study, establish a research quality, reach-scale study to measure the effects of planting and managing a sizeable area (30–50+ hectares) of SRC, ideally across both sides of the floodplain, on flood flows.
- Either separately or as part of one of both of the reach-scale studies outlined above, establish site studies to measure the effects of floodplain woodland on a number of soil and hydrological processes, including water use/evaporation, soil infiltration, soil water storage and soil erosion/sediment delivery. Such a study should also examine how the floodplain woodland affects floodplain hydraulics, energy losses and channel movement/formation, including interactions with LWD and backwater effects.
- Improve the way that coupled hydrology and hydraulic models represent floodplain woodland processes, both in terms of process inclusion (water use/evaporation, soil infiltration, surface roughness and ideally sediment interactions) and the selection of appropriate parameter values. Use data from the site studies to calibrate and test models.
- Use appropriately parameterised and calibrated models based on the data from the site studies to explore the effects of varying floodplain woodland extent, placement, design and management on flood flows.

Chapter 4. Run-off management





4.2 Soil and land management



4.3 Land and headwater drainage management



4.4 Runoff pathway management

4 Run-off management

This chapter forms the literature review for run-off WWNP interventions.

Sections 4.2 to 4.4 cover in detail each of the different WWNP interventions reviewed:

- 4.2 Soil and land management
- 4.3 Land and headwater management
- 4.4 Run-off pathway management

4.1 Introduction

4.1.1 What is WWNP in the context of run-off management?

This chapter reviews how run-off management can deliver flood management benefits. This introduction provides an overview of the hydrological pathways by which rainfall is delivered to rivers during storm events and introduces types of run-off management measures that can influence and manipulate these pathways for flood management purposes.

4.1.2 Delivering quickflow to rivers

The flow (discharge) in a river during a flood event (see Figure 4.1) is made up of 2 main components – baseflow and quickflow. Baseflow is the contribution made by groundwater (Shaw 1983). This section focuses mainly on quickflow (the water generated by a storm rainfall event), which consists of 2 elements:

- Surface run-off the movement of water over a land surface and downslope towards a surface water body
- Throughflow the water that is temporarily stored in the soil

Surface run-off and throughflow can be interconnected when the soil becomes saturated and any additional precipitation causes run-off.

'Effective rainfall' is the proportion of rainfall during a storm that finds its way into a river (Shaw 1983). The remainder is lost though evaporation, surface detention, retention in the subsurface and percolation into the underlying soil/subsoils/geology.

To manage quickflow for flood management purposes involves manipulating the storage in the landscape (soil storage and surface water detention) to slow the rate (travel time) at which water reaches the surface water body. Changes to travel time can be achieved by modifying the:

- throughflow and baseflow inputs through changes in infiltration
- length of run-off flow paths
- roughness of the run-off flow pathways
- landform to reduce or increase the connectivity of run-off flow pathways

The manipulation of throughflow is discussed in Section 4.2; the rest of the review focuses on surface run-off manipulation. The remainder of this section provides the reader with background information to set the context for Sections 4.2 to 4.4.



Figure 4.1 Storm hydrograph elements

Source: Modified from Musy (2001)

4.1.3 Role of ground cover, soil and land drainage

Flow pathways

When rainfall hits the ground surface, it can take different pathways (Figure 4.2) to a water body:

- infiltration following a groundwater flow path
- infiltration followed by throughflow through the unsaturated or saturated zone
- pooling on the surface followed by overland flow

The nature and characteristics of the soil, subsoil and underlying geology, ground cover and land gradient, together with the presence and effectiveness of any artificial underdrainage in the soil and the influence of land management practices, all play an important role in controlling the partitioning of flow through these pathways. The travel time of water in these pathways and the amount of available storage capacity can significantly differ within and between catchments. Hence if WWNP measures can manipulate this component, they could have an impact on the delivery of quickflow to rivers.

Quinn (2012 notes that a typical soil with a vegetated surface will partition the incident rainfall into a number of pathways depending on rainfall duration and intensity, together with the characteristics and moisture status of the vegetation and soil. Three zones are
highlighted (see Figure 4.3); interactions between them can lead to 3 separate flow mechanisms (see Figure 4.4):



Figure 4.2 Flow pathways





Source: Quinn (2012)

Zone 1 defines the infiltration of water into the soil. The near surface infiltration capacity determines the rate at which water is able to enter the soil. When there is excessive rainfall on the surface that overwhelms the soil's capacity for infiltration, water pools and eventually begins to flow across the surface as infiltration-excess overland flow (Figure 4.4A). The infiltration capacity is a function of soil properties and the soil moisture status, and therefore changes during the year and during the storm event itself.

In Zone 2, the water in the soil will generate a local water table and will usually flow under gravity and/or pressure downhill. Emergence of this water further downhill can give rise to

zones of saturation (Dunne and Black 1970) on the toe slopes and generate large amounts of subsurface stormflow (Hewlett and Hibbert 1963) (Figure 4.4B). The zones of saturation can therefore expand and contract, affecting the subsurface flow. Most rainwater falling directly on the fully saturated zone will run-off as saturation-excess overland flow. Saturation-excess overland flow can be a mixture of fresh rainfall and subsurface return flow that emerges at the surface. However, saturation-excess overland flow can also occur for prolonged periods even after rainfall has stopped, as the subsurface flow continues to leave the hillslope at the surface. Therefore the ability of the soil profile to temporarily store incident rainfall by infiltration is controlled by its soil moisture condition, which varies temporally and spatially. Since saturation-excess overland flow is important to overland flow generation in the UK (temperate humid zone), the spatial distribution of subsurface saturation and the times of saturation are critical to understanding the role of WWNP measures.

In Zone 3, the amount of water that reaches the regional groundwater table is found. The vulnerability of Zone 3 to the management of Zone 1 and Zone 2 is vital to the sustainability of the regional groundwater level and the perennial flow of watercourses.



Figure 4.4 Hillslope hydrology response types to incident rainfall events: (A) infiltration-excess; (B) subsurface stormflow and saturation-excess flow where the water table has risen in response to rainfall (dotted blue line); and (C) perched water table causing shallow interflow and water logging or drain flow removing surplus flow

Source: O'Connell et al. (2004)

Controls on flow pathways

The pathways discussed above can be modified by anthropogenic influences as follows.

- In-field underdrains and ditches can suppress the water table and reduce saturation, thereby potentially reducing the occurrence of saturation-excess overland flow (Figure 4.4C).
- Soil degradation, including compaction, can reduce infiltration and thereby increase infiltration-excess overland flow.
- The installation of underdrainage and/or remedial drainage treatments can increase infiltration but also the rates of throughflow.

Soil type, condition, local gradients and vegetative cover are all key determinants of water flow both into and through a soil, and this will determine which rainfall conditions are most likely to induce rapid surface flow. To illustrate the complexity and potential problem encountered in farmed landscapes, Figure 4.5 shows how flow partitioning can be modified. It shows how intensive arable production can cause the soil macro-structure to be degraded and compacted, and thus the need for underdrainage and/or remedial treatment increases. A soil in the condition shown in Figure 4.5 can still produce a good crop yield, but the run-off dynamics have changed and intensive ongoing agronomic management will be needed. Good soil structure is essential for productive and profitable agriculture, and it is difficult to retain good soil structure for agriculture without some form of drainage across many soil types in the UK. A soil can migrate both quickly or progressively in form and function from a non-degraded to degraded condition, but it can take a long time to change back without active intervention.



Figure 4.5 An intensively farmed arable soil with underdrainage, secondary drainage (mole drains), a compacted soil layer and surface run-off flow pathways

Source: O'Connell et al. (2004)

Agricultural underdrainage increases the aeration of the root zone for a crop, assists the soil in warming in spring and can help to reduce the risks of structural damage from machinery and livestock. As such, it is estimated that ~40–50% of the UK landscape has been underdrained with field drains (ADAS 2002). The other 50% is either classed as free draining or as unlikely to be economically drained (ADAS 2002). There have been periods of great drainage activity, for example, in the 19th century. O'Connell et al. (2004) noted that:

'Virtually all fields requiring drainage for effective farming will have some form of drainage installed, and many will have experienced several attempts at underdrainage over the last 250 years'.

The installation of in-field underdrains will generally cause a reduction in overland run-off and near surface flow through an increase in the available storage capacity of the soil column (Figure 4.6). Underdrainage does this through 2 mechanisms:

- direct suppression of the water table
- the removal/interception of water in the unsaturated zone

But although storage is increased, the travel time of flow through drains can also be increased or decreased. Robinson (1990) studied a wide range of experimental plot studies in the UK and deduced that, after underdrainage is installed, peak flow will increase or decrease based significantly on the soil characteristics of the landscape. He showed that drainage of clayey soils tends to decrease peak flow while the drainage of sandy soils tends to increase peak flows.



Figure 4.6 Example of soil drainage concepts used by farmers as presented in guidance from the Agricultural & Horticultural Development Board

Source: AHDB (2015)

When water starts flowing across the soil surface as run-off, the characteristics of the flow will be determined by the local topography and influenced by other land management infrastructure (for example, tracks and roads) and cultivation decisions (for example, tillage and agronomic tramlines). While topography and gradients are substantially fixed in the landscape, decisions on soil management and crop/vegetation management can be varied. As such, these management decisions can affect the physical soil properties and conditions that can exacerbate flood events.

4.1.4 Role of arterial drainage and overland flow routes

Run-off, throughflow and groundwater flow can be intercepted by ditches and drains before it reaches the river. Drainage networks can therefore be crucial in manipulating the delivery of quick flow to a river. Where run-off is not collected by drainage systems (or overwhelms a

drainage system), overland flow routes can be intercepted by well-sited landscape measures to temporarily store, store and filter the quality of the run-off water.

In-field underdrains usually connect to open ditches along field boundaries which, usually through a network of larger ditches and river channels, convey the water across and down through the landscape, eventually to the sea. The depth of underdrains in the soil profile and the water levels in the receiving network of ditches are used to manage localised water tables in the soil. Open surface drains/ditches are also used to intercept surface water run-off and move this water quickly and efficiently from the land.

The role of land drainage and overland flow routes in flood generation is the focus of subsequent sections. In addition to impacts on flooding, they can cause other issues to such as localised muddy floods, problems for infrastructure assets, impacts on local biodiversity and probably contribute to lowland floods (Evans et al. 2004, O'Connell 2004).

There are a range of potential intervention locations. Run-off flow paths could be more effectively trapped in fields and fieldside ditches, thus slowing or storing flow. Riparian areas could have designated buffer zones and infiltration zones imposed on them. Larger channels could also be targeted to slow or store flow. In certain situations, there may be good reasons to reduce agricultural activity for water quality and biodiversity benefits whereby the whole of a field (or groups of fields) is moved to a lower level of farming intensity, or even to no farming at all.

4.1.5 Peatland systems

The discussion above has focused mainly on basic principles and soils consisting primarily of mineral (sand, silt and clay) material, rather than the organic matter in farming landscapes. The function of peat soils and peatland systems influences their hydrological responses to rainfall in a particular way, as discussed below.

Peat covers 9.45% of the land area in the UK and is recognised as providing crucially important ecosystem services (Lindsay et al. 2014a). Undisturbed peat can have a moisture content of greater than 95% (Lindsay et al. 2014b) and is formed of 2 layers (Lindsay et al. 2014c):

- acrotelm a thin surface layer of peat-forming vegetation
- catotelm an inert, permanently waterlogged peat store

UK peatlands are characterised by shallow water tables and a very low permeability at depth, and hence are often zones of saturation-excess overland flow and near surface flow during rainfall events. The surface microtopography of hummocks and hollows with small pools of water, together with the two-layered peat system, helps to form high and stable water tables which preserve the peat from degradation and allow peat-forming vegetation to be maintained. The natural functioning of this hydrological system can be disturbed by a number of different activities including burning, grazing, atmospheric pollution, forestry and drainage. The initial damage can often then be exacerbated by erosion.

Drainage of peatland can occur in a number of contexts. Draining of peat, through a practice known as gripping, was particularly widespread in the uplands from the 1940s to the mid-1980s in an attempt to improve conditions for grazing and game production (Stewart and Lance 1983). This agricultural policy driven drainage activity has subsequently been deemed to have had few positive outcomes with respect to the original objectives. On the contrary, it has now been associated with increased flood risk (Lane 2001), carbon release, water colouration (Worrall et al. 2007a), vegetation loss (Coulson et al. 1990), changes in aquatic communities (Ramchunder et al. 2009), sediment losses and soil pipe development (Holden 2006). Drainage affects the natural functioning of the peatland, creating a feedback loop of

lowered water tables, consolidation, changes in microtopography and vegetation, and oxidative wastage (Figure 4.7). This leads to the loss of the peat body, its reprofiling into a new stable configuration or effects on both water run-off and quality characteristics.



Figure 4.7 Impact of Drainage on Peat Hydrology

Source: Lindsay et al. (2014b)

Upland peat restoration employs a number of techniques to restore the natural functioning of peatland systems. The first priority of restoration is surface stabilisation to stop further erosion and limit wastage (Lindsay et al. 2016). The second priority is to reduce water loss and develop peat-forming vegetation. Following these priorities, many restoration techniques have been developed focusing on the following areas:

• blocking of drainage features

- reducing intentional burning of moorland vegetation
- reducing grazing pressures
- reprofiling of the peat surface
- gully stabilisation mulching and blocking
- revegetation techniques including Sphagnum introduction

As well as the re-establishment of peatland habitats, there are potential multiple benefits that could be derived by peatland restoration, including NFM. Restoration measures have the potential to:

- change the storage of water within the peatland habitats, through rewetting
- change the travel time of water through the system by blocking grips/gullies and other drainage features
- · changes to the roughness of the peatland surface

4.1.6 Synchronicity

Different subcatchments respond to individual rainfall events in different ways due to their differing characteristics and the characteristics of the rainfall events (for example, intensity, duration and track) (Pattison et al. 2014). As you move downstream through a river network, any flood hydrograph in the main channel is increased, with the additional inflow received at every tributary confluence by the combination of the 2 respective flood hydrographs. The magnitude of increase to the resulting combined hydrograph just downstream of each confluence depends on the degree of synchronisation of the 2 individual hydrographs (Figure 4.8).





It is possible for run-off management measures that change the shape and timing of the flood hydrograph in an individual subcatchment to decrease or increase the degree of synchronisation, which could in itself decrease or increase the resulting peak of the combined flood hydrograph. In some circumstances, a smaller and delayed flood peak from one tributary (possibly generated by a WWNP measure) may actually lead to a higher overall flood peak in the main channel if the peak time in the channel now matches that of the tributary. Consequently, further data and evidence of the synchronicity risks and benefits of the implementation of measures need to be identified and analysed.

This synchronisation issue can be explored by the application of a number of catchment modelling techniques (Pattison et al. 2014), which can also help to ascertain whether the targeting of measures in particular subcatchments can, on balance, maximise the potential to desynchronise the flood hydrographs in the most beneficial way in terms of flood mitigation. In any catchment, however, there will always be a risk that for a particular set of circumstances (for example, antecedent catchment conditions or storm characteristics) where the actual flood response that is manifested downstream will not follow the desired effect, as a result of NFM implementation, and a detrimental flood effect will be generated.

4.2 Soil and land management

4.2.1 Introduction

This section looks at how changes in soil and land management can reduce flood risk. As discussed in the Section 4.1, the most important determinants of water flow into and through a soil are:

- soil texture and structural condition
- type, characteristics, and amount of vegetation and surface cover
- moisture status of the soil

Land management practices can increase the amount of surface storage, rate of infiltration and capacity of the soil to store water (Leopold and Maddock 1954, Schwab et al. 1993, Hudson 1995). Slowing run-off, or holding it back on land in the headwaters is largely dependent upon vegetative cover and favourable soil conditions (Environment Agency 2003b).

Since World War II, the drive to increase food production, realised in the Common Agricultural Policy, has markedly changed the landscape in the UK (O'Connell et al. 2007). Farms contain many different soils, which will have been modified by the farming activity. This farming activity has increased more deeply compacted soils due to cultivation practices, increased in-field underdrainage which feeds flow to drains and ditches, and created a network of open drains connecting the hill top to the channel (O'Connell et al. 2007). These factors will induce changes in run-off generation and its delivery to the channel network and, consequently, change the way flooding occurs. Since 2000, the floods impacting England and Wales have reinforced general concerns that changing agricultural practices in the UK may have increased the risk of flooding (Wheater 2006). This concern can be found elsewhere across northern Europe (Bronstert et al. 2002, Pinter et al. 2006, Evrard et al. 2007). Arising from this, there is a requirement to understand the dominant hydrological function of the soil and its relationship to different land management measures.

Much of the recent evidence referred to in this section derives from research on other changes in soil and land management which could additionally influence flood risk (for

example, water quality or agronomic productivity). The precautionary principle could be reasonably applied, suggesting that vulnerable soils exposed to high intensity rainfall can produce both high run-off rates and volumes that cause flooding. O'Connell et al. (2004) suggested that the 'absence of evidence is not evidence of absence', so they proposed that good soil and land use management must play a vital role in WWNP.

The paucity of studies with solid evidence and the ongoing difficulty of gathering evidence using the BACI principle make interpretation problematical. This review includes an assessment of modelling studies that link back to process understanding or strong statistical evidence. Insights into how land management can change soil processes are important to building a greater confidence in how the use of WWNP methods can provide a flood attenuation function.

4.2.2 Understanding the science

Soil and land management are inextricably linked. Soil management is discussed first in this section to provide an understanding of hydraulic properties and how these can be interpreted in terms of flood management functions. Next, land management measures are reviewed and the effect of this on soils and on soil hydraulic properties and consequently for flood management functions.

The potential for soil management to provide a flood attenuation function is determined by a multitude of factors including:

- soil structure, texture, permeability, consistency
- soil drainage and underdrainage
- organic material
- soil moisture conditions

These factors can be affected by antecedent rainfall conditions and climate. They are also hugely varied across the country, as a result of the inherent characteristics of the soil at any particular location and how the soil has been managed in the past and is managed now. As such, this complex inter-linking of factors suggests that local solutions to increase soil water storage cannot be applied universally across the UK landscape to all soils.

At the outset, it should therefore be borne in mind that soils differ from each other in many ways. They can differ in: altitude; aspect; parent material; amount of stone, chalk and organic matter; depth; texture; structure; permeability; consistency; drainage; weather; and the treatment they receive. If there were only 2 variants of each of these 14 factors, $2^{15} = 32,768$ different soils would be possible, and if there were 3 variants, $3^{15} = 14,348,000$ different soils would be possible (MAFF 1970, paragraph 28). This shows that the number of different soils in the UK is very large. A single field may contain more than one kind of soil, each with recognisable properties and behaviour. In general, coarse textured sandy soils will have greater infiltration rates than clay soils, although this can easily be reversed when the surface of sandy loam soils becomes sealed due to crusting and when clay soils dry and crack (Environment Agency 2003b).

Although there are a lot of soil types, there does not appear to be a strong link between soil type and the generation of local surface run-off (O'Connell et al. 2007). However, due to their particular characteristic, some sandy, silty and slowly permeable seasonably wet soils are more susceptible than others to generating local surface run-off (O'Connell et al. 2007).

Soil texture describes the proportion of sand, silt and clay in a soil, which is a property of the geology of the area where the soil was formed. Lighter textured soils, with a higher proportion of sand, tend to be better drained but due to the lower clay content (<18% clay)

are less 'sticky' and more susceptible to mobilisation. Heavier soils, with more than 35% clay content, tend to be more slowly permeable due to smaller pore spaces between smaller clay particles, and more vulnerable to surface run-off generation.

Soil structure describes 'the combination or aggregation of primary soil particles into aggregates or clusters (peds) that are separated from adjoining peds by surfaces of weakness' (Ritzema 1994). Soil porosity is 'the volume of voids as a fraction of the volume of the soil' (Ritzema 1994). Soil structure and porosity are mutually dependent properties of soils; a change in soil structure by improvements or deterioration is associated with a similar change in soil porosity (MAFF 1970, paragraph 78). A well-structured soil allows free movement of water and air between the voids (pore spaces), which is vital for supporting microbial and plant life. The permeability or porosity of a soil is thus affected by its soil structure. Individual particles can be bound together to a less or greater degree into units known as aggregates; the degree of binding is dependent on clay content and organic matter content, and sometimes calcium and iron compounds. Although the Countryside Survey of soils 2007 did not confirm the widespread loss of soil carbon (0–15cm) (carbon can be considered a measure of organic matter), there is a loss of carbon from the intensively managed Arable and Horticulture Broad Habitat / Crops and Weeds Aggregate Vegetation Class (Emmet et al. 2010).

The main properties of soils that allow adequate removal of water by natural drainage are soil depth, texture, structure and porosity, and with shallow soils, the comparable properties of the underlying material (MAFF 1970, paragraph 104). Consequently, the 2 most important factors that influence the flow of water through and out of a soil are permeability and hydraulic gradient. When the soil's capacity to drain is exceeded, water will lie on its surface and, where these soils lie on a slope, the water will move downslope across the soil surface as surface run-off.

There is a spectrum of soil porosity from micropores to macropores (Beven and Germann 1982). Macropore volume tends to be a small proportion of the total pore volume. Well-connected macropores, however, are capable of conducting a significant water flux because of their high transport capacity (Niehoff et al. 2002). In conditions where the soil has little matrix hydraulic conductivity and/or high rainfall intensity, a large part of the infiltration may pass through the macropores into deeper soil layers, bypassing the matrix close to the soil surface. Infiltration into the macropore system occurs only when the net precipitation intensity exceeds the matrix infiltration rate. The influence of macropores on storm run-off generation depends enormously on rainfall characteristics, soil moisture conditions and the connectivity of macropore networks (Niehoff et al. 2002).

The speed at which any soil drains will depend on the soil properties, its moisture content and the position of the soil on the hillslope. Soil wetness therefore varies from location to location. Underdraining is a common agricultural practice in the UK, which uses an underground pipe system to drain soils and improve production (Wheater and Evans 2009). This agricultural practice of underdraining was encouraged by government grants between the 1940s and 1980s (Robinson and Armstrong 1988). The installation of field drains causes a reduction in surface and near surface run-off due to a lowering of the water table and increase in the available storage capacity of the soil (Wheater and Evans 2009). However, the installation of field drains does not necessarily mean that there is an increase of flood attenuation capacity within the soil since run-off from drained land can be faster or slower than from undrained land depending on the properties of the soil and its management (Robinson 1990, Armstrong and Harris 1996), and the timing and intensity of the rainfall (Wheater and Evans 2009). For example, the results obtained by Reid and Parkinson (1984) determined that run-off response from drained fields varied according to the season and the antecedent moisture conditions. Robinson et al. (1985) showed that the installation of field drains in a 16km² clay catchment in north-east England reduced the time to peak flow and increased the magnitude of peak flows.

Additionally, there may be differing short- and long-term effects of draining soils rich in organic matter. For example, Holden et al. (2004) determined that lowering the water table in peatlands will increase the amount of available storage capacity in the short term. In the long term, however, the organic matter decomposition rates will increase and result in a decrease in available storage. Consequently, there may potentially be an increase in flood peaks and soil damage in the long term (Wheater and Evans 2009).

Soil aeration and subsoiling

Soil aeration and subsoiling are common methods to mitigate against artificially induced soil compaction found in arable and grassland systems (from tractors or grazing regimes).

Soil aeration is a process that breaks up topsoil compaction and makes the cultivation of arable crops possible because crops cannot grow in a reduced soil (with the exception of wetland rice) (Ritzema 1994). Targeted soil aeration is a common practice for agronomic reasons, but it can also aid soil function. Mechanical soil aerators include topsoil looseners or aerators. These have been reported to improve the soil physical quality (increase soil macroporosity, hydraulic conductivity, decrease soil bulk density) and infiltration of soils susceptible to livestock damage (Curran Cournane et al. 2011). Breaking up the compaction and increasing the soil hydraulic conductivity is believed to increase soil infiltration and water retention capacity and, consequently, increase the travel time for incident rainfall to reach the arterial drainage system.

Subsoiling (Figure 4.9) is a type of soil aeration, but is likely to be performed to mitigate compaction and increase drainage into the subsoil (below the cultivated layer). Subsoiling is the act of loosening the subsoil with a suitably shaped tine. It is carried out if the impermeable layer of the soil is close enough to the surface and not too thick to break it up and improve drainage and encourage better plant growth (Castle et al. 1984). It is used in the UK on very low permeability soils to improve the flow of water to the underdrains.



Figure 4.9 Equipment for subsoiling and topsoil loosening

Source: AHDB (2015)

The single or double bladed subsoiler is a tool used to improve the permeability in heavy or fine-textured soils. Compared with moling, the emphasis is more on lifting and shattering compact subsoil rather than on forming a stable drainage channel for conducting water (Davies et al. 1972). As such, subsoiling is most often used to relieve localised and particular problems at depth in the soil profile such as compaction, which might be affecting drainage and plant vigour.

A subsoiler should be used at an appropriate depth and spacing, when soil conditions are favourable. It would be sufficient alone where the underlying soil is permeable (that is, it is a free draining sand to depth) (Castle et al. 1984). However, below the impeding layer, if there is a 'true' water table problem that will require controlling, an underdrainage system may be necessary (Castle et al. 1984). If the subsoil has a fine texture, subsoiling alone could well increase the problem by letting surface water pass into the subsoiled area, which becomes waterlogged with consequent deterioration of soil structure and reduced load-bearing capacity (Castle et al. 1984). A system of pipes and subsoiling in low and medium permeability systems will improve the soil permeability from the subsoiling, and allowance can therefore be made for more excess water reaching the drains (Castle et al. 1984). Figure 4.9 shows 2 examples of managing soil structure that targets both a deeper compacted layer (winged subsoiler) and a near surface soil 'loosener' to improve surface drainage and aeration.

Land management

Research points to soil management affecting a soil's potential flood attenuation function. Soil management measures that maximise the potential soil water storage capacity and therefore potentially reduce flood risk will be greatly affected by land management. Soil and land management are inextricably linked. A range of cropping and stock management systems in UK agriculture has the potential to modify soil hydrology (see, for example, Boardman 1991, Boardman 1995, Chambers and Garwood 2000) by affecting soil structural conditions (Figure 4.10) (Holman et al. 2003). There is a lack of quantified data from the UK on river flow response to increase run-off due to soil structural degradation (Holman et al. 2003).



Figure 4.10 Fundamental soil physical properties and surface crusting can affect the relative proportions of overland flow, lateral throughflow and vertical infiltration

Source: Holman et al. (2003)

It is important to appreciate the difference between land use and land management. Land use is a description of how land is utilised, its function or socioeconomic activity (for example, urban, agricultural, industrial, recreation), while land management is the process of managing the use and development of land resources. This literature review focuses on different measures applicable in arable systems, grassland systems and related agricultural landscape features (buffer strips and hedges) in maximising the flood attenuation function in rural landscapes.

Soil structural degradation (due to compaction) is an important factor in run-off generation. By influencing the soil structural conditions that determine the inherent storage capacity within the upper soil layers and their saturated hydraulic conductivity, land management can affect the local generation of surface and subsurface run-off. Management practices that cause soil compaction at the surface reduce the infiltration capacity of the soil and can lead to infiltration-excess overland flow (O'Connell et al. 2007).

Agricultural land use change from arable to a grassland system is considered an option to increase flood attenuation function. However, this change is often driven by biodiversity and other benefits rather than for flood risk management. This change in the land use does not necessarily mean an increase in flood storage, since the management of this grassland system is an important factor in its role in flood risk management, as described below.

The scope of this literature review does not cover land use change. However, land use change is considered in the report by BIO Intelligence Service and HydroLogic (2014) report and the Defra Research Project FD2114 (Environment Agency 2007), which looked at the impacts of rural land use changes and management on flood generation in rural areas at both local and catchment scale within the UK.

In recent studies in south-west England, it was observed that soil structural damage caused by compaction, smearing/slaking and capping reduces vertical water movement through soil by reducing porosity, increasing density, and in severe cases, resulting in the formation of massive structure less layers (Palmer and Smith 2013). Soil texture and the soil water regime have a profound influence on the susceptibility of soils to structural damage (Palmer and Smith 2013). Both arable and grassland farming inevitably change topsoil structure and bulk density, which affects the soil hydraulic properties (Palmer and Smith 2013). The poor timing of cultivation or accessing saturated land with machinery or stock can cause compaction, severely damage soil structure, reduce pore space in topsoils and dramatically change the way water moves over and/or through the soil.

Field investigations between 2002 and 2011 reported by Palmer and Smith (2013) identified soil structural degradation to be widespread in south-west England, with 38% of the 3,243 surveyed sites having sufficiently degraded soil structure to produce observable features of enhanced surface run-off within the landscape. The research by Holman et al. (2003) looked at 5 cropping/management systems identified as having the potential to cause soil degradation:

- autumn-sown crops
- late autumn harvested crops
- field vegetables
- orchards
- grassland both permanent and ley grassland, but not including rough grazing

The results of the interpretation of the field observation data and the extrapolation on a catchment scale showed that severe soil structural degradation in the 4 catchments studied (Severn, Yorkshire Ouse, Uck and Bourne) was associated with late harvested crops such as maize and sugar beet, and at least during the autumn of 2000, potatoes (Holman et al. 2003). High degradation occurred on:

- ~55% of inspected sites with late harvested crops
- 30% of sites under grass, autumn-sown crops and field vegetables

• 10% of sites under orchards

This study did not investigate the effects of overgrazing in upland peaty catchments. The results showed an increased run-off of around 10–20mm from the total area of the 5 land management systems for large rainfall events in the Severn, Ouse and Uck catchments, and around 5mm in the Bourne catchment. However, these results are only likely to occur during exceptional years with prolonged wet weather and cultivation practices leading to widespread soil structural degradation (Holman et al. 2003). Remedial actions to improve soil structure are either not being undertaken or are being used unsuccessfully (Palmer and Smith 2013).

The effects that improved soil management could have on run-off generation and peak flows in the 1,650km² Parrett catchment (55% grass and 45% arable) were investigated using the Soil Conservation Service Curve Number (SCS-CN) approach as the methodology for the study (Environment Agency 2003b). The SCS triangular unit hydrograph assessed both flood timings and peak flows for a 75km² subcatchment with the introduction of good practices in arable and grassland management, which were represented by reductions in the SCS-CN run-off coefficient. The hydrograph peak was predicted to occur 1.5 hours later and with a 20% reduction in peak flow when good practices were applied, for both a one-day event and a series of events over a 5-day period in December 1999, although the total run-off volume over an entire event remained unchanged (Environment Agency 2003b). However, major uncertainties associated with these predictions were not acknowledged (O'Connell et al. 2004).

These studies provide strong evidence for enhanced soil degradation when current cropping and stock management practices have to be performed in less desirable conditions (Holman et al. 2003). It further suggests that a significant amount of run-off entering rivers can increase with enhanced soil degradation, though the realisation of this potential is less certain (Holman et al. 2003).

Arable systems

Agricultural practices that use larger machinery to produce uniformly fine seedbeds for autumn-sown crops and for late harvesting of crops (maize, sugar beet and potatoes) can compact subsoils and weaken the topsoil structural stability of slowly permeable soils with varying amounts of seasonal wetness, through smearing and compression. These practices can cause weakly structured soils or no vegetative cover, which can also lead to infiltration-excess overland flow as a result of the rapid formation of a surface crust with a very low moisture storage capacity and hydraulic conductivity (O'Connell et al. 2007). Increased compaction at the base of a plough layer can also lead to saturation-excess overland flow and a perched water table, leading to subsurface run-off by rapid lateral through flow in the upper soil layers (O'Connell et al. 2007).

Some soils such as sands and light silts have inherently weakly structured soil peds. The production of fine seedbeds in autumn or the late harvesting of crops such as maize, sugar beets, and potatoes on these soils can lead to rapid crusting and 'capping' of the soil surfaces, assisted by raindrop impact, which increases surface run-off (Holman et al. 2003). This evidence was further supported by Sibbensen et al. (1994), who determined that the prevalence of autumn-sown cereals can increase local surface run-off. Speirs and Frost (1985) also found that an increase in maize crops and the production of fine seedbeds can result in an increase in local surface run-off. Palmer and Smith (2013) conducted an extensive survey of soil structural degradation in south-west England under many cropping systems and confirmed its linkage to the generation of enhanced surface water run-off.

These poor agricultural practices can exacerbate the 'normal' response of streams to rainfall and are likely to have the greatest effect during extreme rainfall events at critical times of the year in late autumn, early winter and spring (Holman et al. 2003). A field-based method of soil structure assessment (quick and cheap to conduct) and a field diagnosis of structural damage allows any necessary remedial actions can be identified in less than an hour (Palmer and Smith 2013). The Visual Evaluation of Soil Structure (VESS) is also a simple and quick soil test to assess topsoil structure in 3 simple steps.¹⁷

Remedial actions to increase soil water retention capacity and travel time include:

- conservation tillage
- early sowing and cover crops
- crop rotations

Although there are different measures in arable systems to reduce flood risk by influencing the soil hydraulic properties, they are often conducted together, as exemplified through the experiments and evidence presented below.

Conservation tillage

Conventional agriculture normally relies on the plough as the primary tillage tool, resulting in the most disruptive form of soil tillage, 'inversion' tillage, where the topsoil is turned completely upside down and thus its function can be greatly altered in the short term (Friedrich and Kassam 2012). Soil cultivation or tillage is a mechanical modification of the soil which can, in the short term, have positive effects on soil water retention capacity by decreasing soil bulk density and increasing porosity (BIO Intelligence Service and HydroLogic 2014). In the long term, however, the continuity of macropores is destroyed, disturbing soil structure and rapidly reversing soil water retention capacity (Strudley et al. 2008). The soil microfauna (particularly the earthworm populations through mechanical action or indirectly through disturbance of the soil matter) is also greatly disturbed through tillage (Binet et al. 1997, Paoletti et al. 1998, Lamandé et al. 2003), accelerating the loss of organic matter and potentially reducing the soil water retention capacity (Zhou et al. 2008).

Any reduction in tillage intensity is considered to constitute 'conservation tillage', leading to a wide spectrum of 'minimum tillage' approaches from no tillage and reduced tillage. These approaches leave a minimum of 30% of the soil surface covered with crop residues (Soane et al. 2012). Soil structure tends to be preserved (that is, less heterogeneous) in conservation tillage due to the absence of incorporated crop residues and soil displacement and fragmentation during tillage, and consequently an increase in soil cover, carbon stocks and soil adhesion (BIO Intelligence Service and HydroLogic 2014). Thus, conservation tillage increases the amount of water stable aggregates (Keretsz et al. 2010) and increases macropores connectivity through the action of earthworms, which in turn increase the soil water storage (BIO Intelligence Service and HydroLogic 2014). No tillage farming (also called zero tillage or direct drilling) is a method of growing crops or pasture from year to year without disturbing the soil through tillage. No tillage aims to increase the amount of water that infiltrates into the soil and increase organic matter retention, which is good for soil structure and the cycling of nutrients in the soil. The most powerful benefit of no tillage is improvement in soil biological fertility, making soils more resilient (Soane et al. 2012).

Soane et al. (2012) presented a range of widely reported soil responses anticipated after the introduction of no tillage (within 5 or fewer seasons) (Table 4.1), which may increase soil water retention capacity and increase travel time.

¹⁷ For further information, see <u>https://www.sruc.ac.uk/info/120625/visual_evaluation_of_soil_structure</u>

Table 4.1Soil responses after the introduction of no tillage (within 5 or fewer
seasons)

Advantages	Disadvantages
Increased aggregate stability, especially near surface	Increased bulk density at 0–25cm depth can lead to poor aeration when wet
Increased organic matter content near surface	Increased moisture content near surface in spring in northern regions delaying drilling
Increased vertical and stable pore structure	Reduced soil surface temperature, especially in spring in northern regions delaying drilling
Increased biological activity, especially earthworms (8-fold increase in numbers, 4.6- fold increase in biomass)	Increased acidity near surface
Increased infiltration rate	Increased accumulation of phosphorus near surface with risks of loss in run-off
Increased hydraulic conductivity in subsoil on well-structured soils	
Increased hydraulic conductivity in subsoil on well-structured soils	
Increased soil strength and load-bearing capacity with reduced damage from traffic	

Source: Soane et al. (2012)

Sowing of winter crops and cover crops

By covering the soil surface (for example, by cover crops), soil organic matter can be returned to the soil (crop residue management, organic amendment) being highly beneficial to soil water retention capacity (BIO Intelligence Service and HydroLogic 2014). The type and amount of vegetation and cover protects the soil surface from raindrop splash and capping. It also reduces erosion and increases organic material in the soil profile (increasing evapotranspiration) and maintains open channels for longer (increasing infiltration rate and soil water storage capacity) (Environment Agency 2003b). According to Goeck and Geisler (1989), grass and clover crops can be sown as an 'understorey' or 'intercrop' to row crops such as maize, but can significantly reduce yields.

Crop rotations

These stable all-arable soils merge into those that can be cropped for a time but need a periodic grass break to retain structure (MAFF 1970, paragraph 291) and hence fertility, and potentially having an impact on soil hydraulic properties.

Crop rotations use at least 4 different crop types through a minimum 5-year rotation. Crop rotation or integrated crop management (ICM) (Defra 2017) has been suggested as an improvement to reduce the risk of soil degradation. For example, if a typical 7-year crop rotation that is dominated by cereal crops only has one year in pasture, the soil can be degraded and lead to crop disease. In contrast, a 4-year cycle with one year in grass will lower the farming intensity and should not greatly affect the profitability of the farm; however, it will increase soil hydraulic properties to reduce flood risk for the whole landscape if the benefits of the ICM plan can be achieved (Defra 2017). Appropriate crop rotation practices that prevent the soil from being left bare at critical periods in time can enhance soil water retention capacity. These are often found in extensive systems but can be introduced in more intensive rotations (BIO Intelligence Service and HydroLogic 2014).

Grassland systems

Grasslands cover 69% of global agricultural land (Macleod et al. 2013) and in the UK grassland (permanent and temporary grass) is ~46% of the total UK land area (Defra 2016); therefore, these systems offer great potential to intercept rainfall or modify run-off generation and potentially mitigate flood risk (Macleod et al. 2013).

Grassland systems and livestock production have been greatly influenced as a result of the significant intensification experienced as a direct result of national government and European incentives to increase productivity (O'Connell et al. 2007). Although sheep numbers are currently in decline, the percentage increase in sheep was 142% in England and 181% in Wales between 1950 and 1990 (Fuller and Gough 1999). This increase in sheep numbers modified management practices to increase productivity, including removing hedgerows to enlarge field sizes, developing 'improved grassland' habitats through the installation of field drainage systems and applying fertiliser regular (Marshall et al. 2014). Similar to the measures taken to increase productivity in arable systems, these practices (including grazing management and livestock farming) are believed to lead to:

- an increased risk of flooding as a result of compaction of the soil surface
- a reduction in soil infiltration rates
- an increase in the bulk density of the soil

All lead to an increase in rapid surface run-off from hillslopes into streams and rivers (O'Connell et al. 2007).

Conversely, there is a perception that changes in agricultural management could reduce and/or delay locally generated surface run-off with the potential to significantly reduce downstream flood risk (O'Connell et al. 2007, Wheater and Evans 2009, McIntyre and Marshall 2010).

The literature review by Bilotta et al. (2007) concluded that the species and age of the grazing animal will determine the amount of pressure exerted on the soil. This pressure will affect the soil structural alteration based on the stocking density, soil moisture content, soil texture and the presence/absence of a protective vegetation cover. Cattle exert the greatest forces onto soil, while the breed, age and movement vary the pressure, exerting from 60kPa to 90kPa when static, and leading to forces of up to 400kPa when walking (Bilotta et al. 2007). Sheep have a lower body mass and consequently lower hoof forces than cattle with 50–80kPa while static and up to 200kPa when moving (Bilotta et al. 2007). In general, livestock grazing has an impact on the hydrological cycle because it reduces vegetation cover and consequently interception and evaporation (Samsom 1999), and increases the poaching of wet soils and the compaction of dry soils (Greenwood and McKenzie 2001). The impacts of livestock trampling can include:

- the development of a thin impermeable layer in the topsoil A-horizon (Warren 1986)
- reductions in total porosity, mostly due to a reduction in macropores
- a reduction in root growth due to soil compaction (Greenwood and McKenzie 2001)

Vegetation cover is also another way to manage the impacts of grazing animals (Climo and Richardson 1984, Scholefield and Hall 1986, Kellett 1987). This vegetation cover protects the soil by providing a physical barrier between the hooves and soil (O'Connor 1956), while and the below-ground plant matter (roots and stolons) in the soil act to increase the shear strength of the soil and its load-bearing capacity (Patto et al. 1978). The vegetation cover

can also provide an indirect benefit through the subsequent decomposition of plant residues which bind with the mineral component of the soil and together with other agents, give rise to water stable aggregates, which are more resistant to deformation (Patto et al. 1978). The degree of this protection depends on the quality and quantity of the vegetation (see Section 4.2.5).

Land management has been found to affect local run-off production, although it is uncertain how it will aggregate up to have an impact on whole catchment response and thus flooding (Fowler 2005). There is extremely limited evidence (both qualitative and quantitative) that takes into account the complexity of catchment hydrological connectivity, flood-generating processes and land management across vast areas to determine the type of land management required to create an impact of flood risk on a catchment scale.

Agricultural landscape features

This section discusses hedges and buffer strips, as these are specific agriculturally related features within the landscape that have been modified over the years and have been actively promoted in recent decades for conservation reasons.

Hedges

A hedge is a type of field boundary that can be defined in different ways (Barr and Gillespie 2000). Petit et al. (2003) presented the results of Countryside Survey 2000 relating to the stock of field boundaries in the UK and defined hedges as:

'a more or less continuous line of woody vegetation (no gaps at the bottom) that is or has recently been subject to a regime of cutting'.

Hedges were originally intended to prevent livestock from moving from one area to another and/or used to mark ownership boundaries (Petit et al. 2003).

With a trend to increasing productivity in arable and grassland systems, the removal of hedgerows was accelerated in the 1960s to increase field sizes with the widespread use of tractors and combine harvesters (Harris et al. 2004). The average field in pastoral Somerset was 5.5ha in 1945 and increased to 9.5 ha in 1995. In arable Cambridgeshire, this increase was 6.5ha to 16ha. The hedgerow length in England and Wales in 1945 was 970,000km and had decreased to 400,000 km by 1990 (Harris et al. 2004). Hedge removal can cause or aggravate soil structural problems. Removal can lead to several soil types being amalgamated into one field, causing cultivation and drainage difficulties (MAFF 1970, paragraph 287).

From the early 1980s, ecological and cultural drivers meant that hedgerows started to become important again in the rural landscape and a matter of public concern and objective of government policy (Petit et al. 2003). As a result of hedgerow incentive schemes, many farms began work to restore and manage hedgerows. The results from a survey in 2000 showed that hedgerow restoration and planting now compensated for any neglect and removal, such that between 1990 and 1998, the decline in hedgerow length was halted (Petit et al. 2001). However, Carey et al. (2008) found that the length of 'managed' hedgerows in the UK decreased by 6.2% between 1998 and 2007, with a large proportion of these 'managed' hedges turning into lines of trees and relict hedges due to lack of management.

Hedges have been shown to be cross-slope interceptors which slow down water flows, potentially reducing travel time of quickflow to reach a surface water body through increasing the likelihood of infiltration, interception and evapotranspiration, and optimise the delivery of associated ecosystem services (for example, flood regulation and erosion) (Harris et al. 2004, BIO Intelligence Service and HydroLogic 2014).

Buffer strips

Buffer strips have been a feature of agricultural landscapes as an effect of rough or wet land that is difficult to work or intentional set aside areas resulting from environmental legislation or compensation schemes (Stutter et al. 2012). Buffer strips are areas of natural vegetation cover (such as grass, bushes or trees) sited in riparian zones, or away from water bodies as field margins, in-field machinery turning areas (headlands) or within fields (for example, beetle banks). They can have several different configurations of vegetation, varying from simply grass to combinations of grass, trees, and shrubs (Stutter et al. 2012). Their primary role includes trapping nutrients and sediments, while riparian buffers can provide multiple benefits in terms of biodiversity and water regulation (Stutter et al. 2012). Due to their permanent vegetation, buffer strips offer good conditions for (BIO Intelligence Service and HydroLogic 2014):

- reducing the connectivity of surface run-off
- effective water infiltration
- promoting the natural retention of the soil, with potential consequences to increase soil water storage and travel time

The 'Rural Sustainable Drainage Systems Manual' (Environment Agency 2012) divides buffer strips into 3 categories:

- dry grass buffer/filter strip: broad, gently sloping area of grass or other dense vegetation that can be placed on slopes around the farm to intercept run-off around vulnerable areas
- riparian buffer strips (dry): medium width, dry bands of natural or naturalised vegetation situated alongside waterbodies
- riparian buffer strips (wet): a broad strip of natural or naturalised wetland vegetation or wet woodland alongside a water body

Buffer strips, similar to hedges, increase the hydraulic roughness of the landscape. This will reduce surface flow velocity, increasing nutrients and sediment deposition prior to their export to the waterbodies (Dillaha et al. 1986). Riparian vegetation represents the ultimate buffer for surface and subsurface run-off from the land to the draining waters (Vought et al. 1995).

4.2.3 Confidence in the science

The confidence in scientific evidence on the impact of land management on soil water retention capacity and travel time is limited. There is confidence in the soil science and the qualitative impacts of land management. However, there is limited quantitative measured and monitored evidence of how land management reduces throughflow and increases water travel times in the UK. At the local scale, there is substantial evidence that changes in soil and land management practices affect run-off generally, but the effects are complex. Although there are local changes in run-off, there is very limited evidence that these local changes are transferred to the surface water network and propagate downstream. The literature provides evidence of soil and land management measures effects on different soil properties (for example, structure, infiltration capacity and soil organic matter), but no evidence was found during this literature review which indicated a direct relationship between these soil properties and flood risk downstream. Teasing out cause and effect in complex drainage basins containing multiple non-stationary processes is difficult. It is made more challenging because rainfall and river flow records are sometimes unreliable and may be relatively short with few extreme flow events to consider (Lane 2017).

Fowler (2005) stated that it is unlikely that land management in itself will provide a robust solution to the flood problems with an increasing frequency and magnitude of extreme rainfall events, particularly given climate change projections. However, there is a lack of evidence about the upscaled impacts of soil and land management measures upon which to base this assessment. BIO Intelligence Service and HydroLogic (2014) conducted an extensive review of studies and literature of behalf of the European Commission which considered the impacts of land use and management on soil water retention capacity while Robinson (1990) conducted a comprehensive and extensive review of field and catchment studies relating to the impact of land drainage on flooding downstream. Robinson (1990) concluded that, in general, peak flows were reduced by land drainage at wetter sites (high rainfall and/or high clay content) and peaks were increased at drier sites (lower rainfall and/or more permeable soils). It was further suggested that the likely effect of artificial drainage (to worsen or reduce flooding) is highly localised and that it can be assessed from the pre-drained response and measurable site characteristics (Robinson 1990). These reviews of the science and literature indicate the variability of the impacts of land management dependent on soil characteristics, soil water regime, land drainage (natural and artificial) and the rainfall regime.

Soil aeration and subsoiling

There is high confidence that soil aeration and subsoiling does increase the ability for soils to infiltrate water and drain, and potentially increase soil water storage capacity, but there is currently low confidence as a measure in itself to significantly reduce flood risk. Additionally, there are limited studies in the UK and consequently there is a reliance on international literature.

In the UK, soil aeration and subsoiling is common practice in arable and grassland systems for drainage and preparatory cultivation for certain crops (that is, sugar beet) (Zhang et al. 2016). The Demonstration Test Catchment (DTC) project surveyed farmers to determine the commonality of subsoiling to alleviate soil compaction and improve soil drainage. Drained arable land (Wensum DTC) was subsoiled much more than drained grassland (Eden DTC); 80% of farmers used subsoiling on drained arable land compared with 35% on drained grassland. The majority of grassland farmers who subsoiled to improve drainage last carried out subsoiling in the previous 3 years, with the majority of farmers, whether arable or grassland, undertaking some subsoiling in the last 3 years (Zhang et al. 2016). Although it is a common practice, the benefits of soil aeration and subsoiling varies and is dependent on soil type and degree of loosening.

Douglas et al. (1998) outlined the beneficial effects of soil aeration on the structural properties of the soil of an increase in the volume, size and number of macropores in the uppermost 100mm that affected infiltration rate, soil strength and accumulation of organic material. These results were supported by experiments situated on 2.5ha of land, 10km south of Edinburgh, on clay loam topsoil in a profile described as imperfectly drained (Douglas et al. 1998). Burgess et al. (2000) and Drewery et al. (2000) in New Zealand have concluded that shallow mechanical aeration on silt loam soil should be conducted on compacted soils with macroporosity values <10% (volume to volume) to increase macroporosity and saturated hydraulic conductivity, and decrease soil bulk density (Curran Cournane et al. 2011). Both studies found that the soil physical conditions reverted back towards those of the control plots within 10 months (Burgess et al. 2000) and 2.5 years (Drewery et al. 2000).

With regard to run-off volumes, the evidence is mixed. Franklin et al. (2007) found a decreased run-off volume for well-drained soils in Georgia, USA, when comparing non-aerated and slit aerated soils (tines pushed into the soil to make elongated holes). Decreases in volumes were attributed to increased infiltration of rainfall (Curran Cournane et

al. 2011). However, in poorly drained soils, soil aeration increased run-off volumes compared with a non-aerated soil treatment (Franklin et al. 2007).

The study conducted by Curran Cournane et al. (2011) tested the hypothesis of whether soil aeration would alleviate soil physical quality, decreasing phosphorus and suspended solids, in poorly structure soils (silt loam soil) in New Zealand. However, the study concluded that there was no significant difference between surface run-off volumes between the aerated and control treatments. It is believed that any changes in soil physical properties (that is, decreased losses in macroporosity) were short-lived and therefore unlikely to influence surface run-off in the long-term for this poorly structured soil (Curran Cournane et al. 2011). These findings cannot be transferred to other sites where infiltration-excess overland flow is the dominant run-off process, but can be applied to sites where surface run-off is generated under saturated-excess conditions (Curran Cournane et al. 2011).

Arable systems

Conservation tillage

Although tillage is the most widely research management practice affecting soil hydraulic properties and processes in the fields (Green et al. 2003), Strudley et al. (2008) showed that there has been a rapid decline of publications into the 21st century with a peak rate of publications in the 1990s in the area of spatial and temporal variability in soil properties due to tillage. Consequently, there is little recent evidence and it is necessary to rely mainly on older evidence. The peer-reviewed evidence is conflicting: it is based on experimental results from field and laboratory studies that illustrate the impact of conservation, minimum tillage and no tillage on run-off, infiltration, and soil water storage (soil hydraulic properties).

Deasy et al. (2009) investigated mitigation options for diffuse pollution losses from arable lands. However, as a consequence of monitoring diffuse pollution, surface run-off was also monitored. The study showed minimum tillage was significantly effective (p < 0.05) in 2 out of the 5 site years trialled (winters of monitoring) in reducing losses of run-off, suspended solids and total phosphorus. This reduction of run-off was found in year 2 on the heavy clay soils at Loddington in Leicestershire (annual rainfall of 650mm) and year 3 on the sandy soils at Old Hattons in Staffordshire (annual rainfall of 700mm). In year 2 at Loddington, overwinter run-off was 31.3mm for minimum tillage compared with 45.3mm for ploughed areas. In year 3 at Old Hattons, minimum tillage significantly reduced losses of run-off, suspended sediment, and phosphorus from tramlines by 66% to 98% (p < 0.01). In other years where minimum tillage was not effective, losses of run-off and total phosphorus were significantly greater (p < 0.05) in areas under minimum tillage than from traditionally ploughed areas (Deasy et al. 2009).

Another study by Deasy et al. (2014) looked at comparing traditional ploughing to minimum tillage at an arable farm with heavy clay soils at Loddington in Leicestershire (Gleyic Camisol) with slopes of ~4° and average annual rainfall of 650mm. The experimental set-up of plots is shown in Figure 4.11. The results were collected from 20 separate rainfall events occurring from November 2007 to April 2008 (average event during this time was 12.9mm), a shorter time period of data collection than for the study by Deasy et al. (2009). The results from Deasy et al. (2014) conflict with some of the Deasy et al. (2009) results, and show that minimum tillage increases the total surface run-off generation (QTotal), peak flow (QPeak) and run-off responses (QDuration) to rainfall events, supported by model results. However, the QLag (lag time between peak run-off response and onset of event rainfall) is increased compared with ploughing, and this result was not supported by the model results (Deasy et al. 2014). This study suggested that run-off may take longer to peak due to the effect of stubbles and crop residues increasing larger scale surface roughness and slowing down run-off transfer to the base of the hillslope and consequently reducing soil erosion under reduced

cultivation techniques (Pierzynski et al. 2000). These conflicting results illustrate a low confidence in how conservation tillage can reduce flood risk.



Figure 4.11 Plot trial layout at Loddington

Source: Deasy et al. (2014)

The international literature also has conflicting results on the impacts of different tillage types and emphasises different outcomes depending on soil characteristics, climate variability and time of evidence. The review of literature in northern, western and south-western Europe by Soane et al. (2012) concluded that the soil and climate types within and between these regions in Europe exert a strong influence on the success of no tillage, along with the handling of surface residues, weed control, compaction control, and the correct selection and use of herbicides and direct bills.

Strudley et al. (2008) provided a detailed synopsis of international literature for quantifying conservation tillage on soil hydraulic properties with particular emphasis on space-time variability, which may be a factor in why there are conflicting results. The analysis demonstrated that experimental results from field and laboratory studies do not support the same trends of conservation tillage on soil hydraulic properties. For example, comparisons of no tillage with conventional tillage practices produced mixed results across all studies. The studies reviewed by Strudley et al. (2008), which observed a trend, showed that no tillage practices increased macropore connectivity while generating inconsistent responses in total porosity and soil bulk density compared with conventional tillage practices. The extensive literature on agricultural management effects demonstrates an awareness of interactions between management practices, other factors of crop residue management, compaction, and irrigation effects, and the complexity of spatial and temporal variability (Strudley et al. 2008).

Early sowing and cover crops

Limited peer-reviewed literature was found during this review, which suggests a low confidence in the method of early sowing and cover crops in reducing run-off, increasing soil water retention capacity and travel time to reduce flood risk. With regard to early sowing and cover crops specifically, the results described in Section 4.2.2 show conflicting results, with very limited UK studies (Environment Agency 2002a). The majority of research on early sowing and cover crops can be found in the grey literature produced by organisations such as:

- European Commission (BIO Intelligence Service and HydroLogic 2014)
- Environment Agency (Environment Agency 2002a, Environment Agency 2003b, O'Connell et al. 2004, Environment Agency 2008a)

Such organisations look to promote sustainable agricultural practices in which early sowing and cover crops can reduce erosion and compaction, but have conflicting conclusions as to the reduction of field surface run-off.

However, based on the principles on soil surface roughness, there is a greater confidence that planting cover crops will increase travel time and infiltration rates. Soil surface roughness is a critical parameter for soil erosion and run-off processes (Zheng et al. 2012). A higher degree of soil surface roughness is thought to enhance the infiltration rate of water through the soil surface (Hansen et al. 1999, Kamphorst et al. 2000, Planchon et al. 2001) and reduce overland flow (Zheng et al. 2012). The hydraulic roughness coefficient quantifies the effect of the soil surface roughness on overland flow. Precise estimation of the hydraulic roughness coefficient is critical to simulation of the processes of overland flow and soil erosion (Zhang et al. 2010). However, the quantifiable impact of increasing soil surface roughness, particularly through cover crops, is not well-documented in peer-reviewed evidence, as it depends on parameters such as flow rate, flow regime, soil concentration, topography and tillage practices (Zheng et al. 2012).

Crop rotations

As yet there is no measured evidence in the UK that changes in crop rotation affect the risk of flooding at the catchment scale. However, on a local scale, crop rotations are often a part of integrated farm management. Leopold and Maddock (1954) stated that improved management including crop rotations, the sequence of planting crops, the use of mulches and other practices that improve the soil tilth were more effective in reducing storm run-off than practices such as terracing, strip cropping and contour cultivation.

There is extremely limited peer-reviewed literature on the impacts of crop rotations on the risk of flooding and consequently low confidence in this management measure. There is also limited knowledge or research found in grey literature, with just a few mentions in Harris et al. (2004), O'Connell et al. (2004) and the update of the O'Connell et al. (2004) report by the Environment Agency (2007). Both O'Connell et al. (2004) and Environment Agency (2007) stated that agricultural crop cycles are not well represented in the modelling of infiltration processes, run-off generation mechanisms and channel processes.

Grassland systems

There is a limited number of observations from targeted, controlled experiments on the impacts of stocking/destocking on run-off generation. The majority of the evidence is qualitative, particularly for upland areas and requires further experiments (Carroll et al. 2004b, O'Connell et al. 2007, Wheater and Evans 2009). The review conducted by Harris et al. (2004) also concluded that there are few UK references relating to grassland management and run-off, though the review did contain international references relevant to the UK's climate, cropping patterns and practices.

The UK references that are available offer a mixed interpretations of results. In ungrazed fields, Heathwaite et al. (1989, 1990) found that there was a reduction of rainfall converted to run-off and increased infiltration capacity compared with grazed fields. Further details of this study are found in Section 4.2.5. Additionally, Lane (2001) suggested a link between land use change within the Ouse catchment (increasing stocking density of sheep during the 1970s and 1980s in the Yorkshire Dales), particularly in upland areas, increased the speed with which rain reaches the drainage network and consequently increasing the frequency and severity of flood events in York. However, Fowler (2005) suggested that the change in flood response could be attributed to changes in rainfall seasonality and an increase in extreme rainfall events rather than grazing pressure.

A UK study on grassland management at Pontbren in mid Wales did not find a significant difference between the soil infiltration rates between ungrazed and the control treatments (grazed plots) between 2005 and 2012 (Marshall et al. 2014). The farmers at Pontbren sacrificed the least productive areas to tree copses and non-grazed areas, providing an ideal opportunity to determine whether the infiltration rates in the copses and the non-grazed areas of the farms would recover and cause any upslope run-off to infiltrate (Environment Agency 2003b). Marshall et al (2014) showed that, prior to treatment, all plots appeared to have a similar run-off response rate with the main difference being the control showing a longer delay in response and slightly larger peak flow rate. Following the treatment and over 2 years of data collection, it was concluded that the hydrographs separated and the control plot had the shortest time to peak and the largest surface run-off volume. The ungrazed plot had a shallower rising limb, smaller peak and a smaller run-off volume. There was no significant difference at the 95% confidence level (p < 0.05) in the soil infiltration rates and soil bulk density (median treatment near surface of 2-5cm depth) between the ungrazed and the control treatments (Marshall et al. 2014).¹⁸ There were some cases in which the natural variability of run-off and infiltration rates was greater than the changes observed following treatment (Marshall et al. 2014), likely to be due to differences in microtopography between plots and spatial variability in soil physical properties (Biggar and Neilsen 1976, Beven et al. 1993).

Although the results from Pontbren do not show significant results between grazed and ungrazed land, it is noted that the rate of change and overall extent of soil recovery depends on a multitude of different factors including soil type, severity of grazing, climate and the presence of biological agents (Greenwood et al. 1997). More rapid recovery is generally observed in temperate climates (Greenwood and McKenzie 2001).

Agricultural landscape features

Hedges

There is very little quantitative or qualitative evidence in the UK on how the management or creation of hedgerows may increase water travel times or soil water retention capacity to reduce flood risk. Discussion of the role of hedges in reducing flood risk is mostly found in the grey literature (Harris et al. 2004, BIO Intelligence Service and HydroLogic 2014).

Dewald et al. (1996) reported on the research by the US Department of Agriculture's Agricultural Research Service, the National Resources Conservation Service and several universities to study the use of grass hedges to control run-off and reduce soil loss. This study found that grass hedges provide an effective and economic way of slowing run-off water reducing soil losses through erosion. Grass hedges differ from other common types of grass strips (that is, buffer strips and filter strips) because they are narrow, planted with stiff

¹⁸ There was a significant difference of bulk density (12–15cm depth) and soil infiltration rates found between the tree (woodland) and control treatments (Marshall et al. 2014). Further information can be found in Section 3.2.8.

and erect grasses, and form a dense uniform stand of coarse stems that slows concentrated run-off, causing backwater as deep as 30–40cm and allowing time for deposition of eroded sediments. The deposited sediments fill in low spots in fields so that future run-off is more broadly dispersed, less erosive and reduces travel time (Dewald et al. 1996).

Similar to vegetative cover and cover crops, there is more confidence that the introduction or preservation of hedges (rather than their removal) will increase the hydraulic roughness of the landscape and reduce surface flow velocity, increasing travel time, soil infiltration and capacity (Hansen et al. 1999, Kamphorst et al. 2000, Planchon et al. 2001, Zheng et al. 2012). The effectiveness for flood flows is not well-documented.

Buffer strips

Although buffer strips have been part of the agricultural system for decades, the scientific literature concerning them remains primarily biased towards single issues (Stutter et al. 2012) such as sediment trapping (Liu et al. 2008, Gumiere et al. 2011) or pesticide retention (Arora et al. 2010); reducing flood risk is not one of them.

The literature regarding the effect of buffer strips on reducing flood risk by increasing soil permeability, soil organic matter or water storage capacity is often a result of information gathered as part of single issue studies (Borin et al. 2010). Buffer strips probably increase the hydraulic roughness of the landscape and reduce surface flow velocity, increasing travel time, soil infiltration and soil capacity (Hansen et al. 1999, Kamphorst et al. 2000, Planchon et al. 2001, Zheng et al. 2012), but there are minimal data on the effects of buffer strips on run-off and soil hydraulic properties. The modelling study by Gao et al. (2016), albeit for upland peat basins, supports the limited evidence base that buffer strips can reduce run-off and suggests different types of buffer strips can have different effects.

Catchment scale

Most of the evidence (described above) consists of local scale studies on the effects of land/soil management on flood risk. At the catchment scale, the impacts are highly uncertain and spatially and temporally dependent (Pattison and Lane 2012). It is the relative timings of the contribution of each subcatchment to the main channel that influences the volume of water at a given location at a given time. The phasing of tributary peak flow with respect to the main channel is a crucial control on how local scale run-off changes are upscaled to the catchment outlet (Pattison and Lane 2012).

Rural land management measures seeking to reduce downstream flood risk need to take into account the fact that the timing of run-off generation and consequently tributary interactions are also driven by the way in which a weather pattern moves across a river catchment (Pattison and Lane 2012). Additionally, the impact on downstream peak flows of an area of compacted agricultural land depends on where in the catchment the changes occur. It also must be considered that the reduction to downstream flood risk at one scale may change for other scales (Pattison and Lane 2012, Lane 2017). Determining the impacts of increasing flow attenuation in one tributary depends on the tributary's relationship with water delivered from other tributaries (Lane 2017). Consequently, determining whether land management will have an impact downstream is strongly scale dependent (Lane 2017) and also based on the interactions between the various factors conceptualised in Figure 4.12. Decisions to recommend land management to reduce flood risk on a catchment scale should only follow from understanding how those mechanisms combine with places to reduce downstream flood risk (Pattison and Lane 2012).

The relationship between land management (for example, arable systems and grassland systems), its practices (for example, crop rotation, timing and livestock density) and downstream flood risk can be determined through a range of processes:

• rainfall partitioning effects

- storage
- run-off timing effects and routing effects
- upstream conveyance and local conveyance (Pattison and Lane 2012, Lane 2008)

As results are upscaled, the confidence in the effect of the soil or land management measure decreases because the increase in spatial scale of interest increases the probability of confounding processes (Lane 2017).

Lastly, catchment-scale modelling of soil and land management measures has a number of challenges (Lane 2017). Once modelling covers large catchment areas, data demands may become excessive and the number and combination of interventions required will increase. Therefore, identifying the optimal sets of interventions due to network effects may be difficult without multiple model runs that are likely to be computationally very expensive at present (Lane 2017). Most prominently, hydrological science is now well-accepted as an uncertain science and this uncertainty may be potentially greater than the change associated with an implementation of WWNP measures (Lane 2017).



Figure 4.12 Conceptualisation of the flood system that links land management changes to changes in flood risk

Source: Adapted from Pattison and Lane (2012), Lane (2008)

4.2.4 Metrics of cost–benefit and multiple benefits

Poor soil structure and enhanced run-off caused by intensive agricultural and livestock practices can potentially mobilise large amounts of sediment and colloidal material (including

soil, plant and livestock faecal matter) from the damaged and exposed soil surface, with this matter entering surface waters where it could contribute to:

- sedimentation problems (Harrod and Theurer 2002, Walling et al. 2003)
- eutrophication (Haygarth and Jarvis 1999, Heathwaite and Johnes 1996)
- pathogenic contamination (Chadwick and Chen 2002, Oliver et al. 2005)

The agricultural sector is recognised as producing substantial ecosystem service flows, particularly via provisioning and regulating services. Consequently, policies, such as agrienvironmental schemes acknowledge this by offering a range of incentives to encourage delivery of these environmental outcomes (Beharry-Borg et al. 2013; 47% of farmland in England was covered by the entry level (Environmental) Stewardship scheme in 2007 (Hodge and Reader 2010). These schemes include grassland management and boundary management (that is, management of hedges). A flat rate environmental payment is triggered once a specified total environmental points target has been achieved (Natural England 2010a). The scope of this literature review restricts an extensive coverage of these agri-environmental schemes (also known as a form of 'payments for ecosystem services'), but references include Morris and Potter (1995) and Kleijn et al. (2006).

Some costings of each of the measures are discussed at the end of this section and the multiple benefits are explored further.

A number of the measures presented in this literature review have other benefits such as reducing soil erosion, diffuse pollution and increasing biodiversity in mitigation against the risks outlined above. Indeed, the additional benefit now being recognised is the potential flood risk benefit through changes in soil properties such as soil storage, water retention capacity, infiltration rates and organic content.

Soil aeration and subsoiling

As discussed in Section 4.2.2, the aim of subsoiling is to improve the soil structure – mitigating against soil compaction – to improve the drainage status of the profile and encourage better plant growth (Castle et al. 1984). Soil aeration can also mitigate against plant diseases since deficient oxygen is a causal factor of plant diseases (Grable 1966). Subsoiling, specifically, can be beneficial in 3 situations to improve field drainage and potentially increase crop yields (Castle et al. 1984). These are:

- compaction pans soil compactions from implements including use of heavy farm machinery when soil is wet, during cultivations, continued use of rotovators and intensive grazing on grassland
- chemical or natural pans downward movement of clay particles and/or iron and humus in acid solution to create a layer of high clay content or an iron pan
- · subsoils with low permeability

Arable systems

Conservation tillage

The initial purpose of conservation tillage is to improve soil structure and stability by protecting soils from erosion and compaction (Holland 2004, Lal et al. 2007) and increase crop yields. In Europe, the area cultivated using minimum tillage is increasing primarily in an effort to reduce production costs, but also to prevent soil erosion and retain soil moisture (Holland 2004).

Conservation tillage can have multiple benefits (Holland 2004). These include:

- reducing the risk of run-off
- reducing the risk of pollution of surface waters with sediments, pesticides and nutrients
- lowering the energy consumption of soil cultivation and the emission of carbon dioxide
- carbon sequestration through the increase in soil organic matter
- improving nutrient recycling (can help combat crop pests and diseases)
- improved food supplies for insects, birds and small animals from crop residues

Holland (2004) reviewed the multiple benefits of conservation tillage outlined above, but concluded that detailed information from European studies is sparse and disparate. With no detailed studies conducted at the catchment scale in Europe, some findings need to be treated with caution until they are verified at a larger scale and for a greater range of climatic, cropping and soil conditions (Holland 2004). Soane et al. (2012) reviewed literature in Europe, compiling a list of advantages and disadvantages of no tillage (Table 4.2).

Table 4.2Relative agronomic advantages and disadvantages of no tillage in
Europe

Advantages	Disadvantages
Lack of compaction below plough furrow	Crop establishment problems during very wet or very dry spells
High work rates and area capability	Weed control problems
Increased bearing capacity and trafficability	Cost of herbicides, herbicide resistance
Reduced erosion, run-off and loss of particulate phosphorus	Risks of increased nitrous oxide emissions and increased dissolved reactive phosphorous leaching
Opportunity to increase area of autumn- sown crops	Reduced reliability of crop yields, especially in wet seasons
Stones not brought to the surface	Unsuited to poorly structured sandy soils
Drilling phased to take advantage of favourable weather conditions	Unsuited to poorly drained soils
Increased area capability	Risk of topsoil compaction
Reduced overall costs (fuel and machinery)	Problems with residual plough pans
	Increased slug damage
	Unsuited for incorporation of solid animal manures

Notes: Not universally relevant to all regions Source: Soane et al. (2012)

Christian and Ball (1994) reviewed crop yields under no tillage in the UK. They suggested that no tillage was capable of providing similar yields to conventional ploughing. Crop yields were appreciably lower after immediate adoption of no tillage, but improved after about 3 years of no tillage as soil structural conditions improved. In the UK, soils with imperfect drainage and weak structure generally led to lower yields with no tillage than after ploughing, especially for spring-sown barley after wet winters.

Cannell et al. (1978) categorised the soils of the UK into 3 classes according to their perceived suitability for no tillage and a map was produced showing their distribution. Throughout the UK, good internal drainage was deemed a prerequisite for reliable success with no tillage (Soane et al. 2012). On underdrained clay soil in England, the no tillage yield of oats was 3.4 tonnes per hectare compared with 6.1 tonnes per hectare after ploughing. The corresponding yields on drained plots were 7.0 and 7.2 tonnes per hectare respectively (Soane et al. 2012).

Deasy et al. (2009) showed that minimum tillage would result in an increase in the farm margin of £44 to £50 per hectare, notwithstanding the capital cost required for equipment and crop residue incorporation. With its high cost savings resulting from reduced operations, minimum tillage may be an attractive option to farmers where it can be implemented, and is normal practice on the heavy clay soils at Loddington (Deasy et al. 2009). There was no evidence from this study of any impact on crop yields by converting to minimum tillage from traditional ploughing. There was also no requirement for increased agrochemical use during the 3-year study, which might occur over time and reduce cost savings (Deasy et al. 2009).

Early sowing and cover crops

Typically, cover crops are non-cash crops sown in the autumn to provide winter ground cover when the field would otherwise be fallow, thereby reducing the risk of soil nutrient losses from leaching and erosion (Cooper et al. 2017). They have primarily been used to minimise nitrate leaching by scavenging highly soluble residual soil nitrate and converting it into a relatively immobile organic nitrogen (Premrov et al. 2014).

The Wensum DTC addressed the issues of assessing the impacts of cover crops and noninversion tillage regimes at the farm scale.¹⁹ The Wensum DTC results discussed in Cooper et al. (2017) focused on impacts on pollution. The key findings can be summarised as follows.

- A winter oilseed radish cover crop reduced nitrate-nitrogen (NO₃-N) leaching losses by 75–97% relative to fallow land, but had no impact on phosphorus losses.
- Direct drilling and shallow non-inversion tillage were ineffective at reducing soil water NO₃-N and phosphorus concentrations relative to conventional ploughing.
- Soil NO₃-N concentrations were reduced by ~77% at a depth of 60–90cm beneath the cover crop, highlighting the potential of long rooting oilseed radish to scavenge nutrients from deep within the soil profile.
- Despite covering 20% of the catchment, improvements in river water quality downstream of the trial area were not observed, suggesting that prolonged use of cover crops may be required before catchment-scale impacts are detected.

Cooper et al. (2017) recommended further research into the effectiveness of other cover crop varieties and crop mixtures at reducing arable nutrient losses, particularly in the UK.

Macdonald et al. (2005) tested a number of cover crop species – forage rape, rye, white mustard, a rye/white mustard mixture, Phacelia and ryegrass. They concluded that early sown cover crops are most likely to be effective when grown on freely drained sandy soils where the risk of nitrate leaching is greatest. They are less likely to be effective on poorer drained, medium heavy textured soils in the driest parts of south-east England. The regeneration of weeds and cereal volunteers, together with some additional broadcast seed, may be sufficient in the driest areas to avoid excessive nitrate losses (Macdonald et al. 2015).

¹⁹ <u>http://wensumalliance.org.uk</u>

There are some associated costs recognised by farmers in growing crops that add an extra expense, complexity and uncertainty (Weil and Kremen 2007). Cover crops have previously interfered with crop production by:

- using up water stored in the soil profile
- immobilising nitrogen needed for the cash crop
- becoming weedy or producing excessive residues hampering crop stand establishment or harvest

The most direct costs associated with cover crops include those for cover crop seeds, labour, fuel, fertiliser, and herbicide or tillage to kill the cover crop (Weil and Kremen 2007).

Crop rotations

Crop rotation has been an important agro-ecosystem management practice conducted to preserve and improve sustainability. The purpose is to increase the system productivity by (Bullock 1992, Studdert and Echeverria 2000):

- affecting nutrient and water availability
- pests, weed and disease dynamics
- presence of growth inhibiting or promoting substances in the soil
- the soil condition

The impact of crop rotations on disrupting disease and arthropod cycles is well-established (Francis and Clegg 1990). Dick (1992) identified a number of studies which determine how crop rotation affects soil biodiversity and its impact on disease interactions, nutrient cycles and crop yields. There are also higher bacterial counts found in a continuous corn or wheat field than fields with a crop rotation (corn–oat–wheat–red clover) in the presence of nitrogen, phosphorous and potassium applications or animal manure (Dick 1992).

Furthermore, crop rotation tends to result in a higher crop yield than continuous monoculturing of a single crop species (Dick 1992). Diverse crop rotations spread the workload, reduce the risk of poor income and minimise the impact of any one crop on the environment (Harris et al. 2004). The inclusion of spring-sown crops in the rotation can bring extra benefits because where stubbles are leftover in the winter can provide cover and food for farmland birds (Harris et al. 2004).

Grassland systems

Destocking or reducing the intensity of grazing has a multifunctional role in producing food and rehabilitating grasslands, and in environmental management and cultural heritage by reducing soil compaction and consequently having an impact on the soil storage, water retention capacity, infiltration, organic matter and surface roughness, and potentially drainage (Kemp and Michalk 2007). The relationship between grazing, productivity and environmental changes will determine the benefits that can be achieved.

Agricultural landscape features

Hedges

The planting, conservation and management of hedges helps to intercept overland flow across slopes in erosion vulnerable areas and reduce the concentration of animal or machinery operations in these vulnerable areas (Environment Agency 2012). Hedges can improve infiltration and sedimentation, retaining eroded particles carrying pesticides and

phosphorus. The soils close to hedges may be oxygen depleted and consequently the hedges support denitrification and trees in hedges may selectively absorb some dissolved elements (Environment Agency2012a).

A major driver for the planting, conservation and management of hedgerows in the UK is biodiversity. Hedgerows provide wildlife corridors (for example, for bats). They are a habitat provision, providing physical shelter and roost sites, and an important source of winder food supplies (particularly berries and other fruits) (Benton et al. 2003, Environment Agency 2012). The effectiveness of the hedgerows in increasing biodiversity depends on the size (height/width/volume) and structural complexity of the hedgerows (Benton et al. 2003).

Oreszyczyn and Lane (2000) investigated the meaning of hedgerows in the English landscape from the perspective of different stakeholders to obtain a holistic view of people's relationship with hedgerows and each other. The study found that, although there was high commonality in the way stakeholders viewed hedgerows in the landscape, differences were found in relation to management practices and their implications for the wildlife and aesthetics.

The review by Baudry et al. (2000) considered hedgerows from an international perspective on their origin, function and management. The study looked at:

- the primary function of hedgerows in the landscape
- as products
- their shifting functions
- current protection, including for the conservation of biodiversity

Although the traditional use of hedgerows as an important source of wood and other products is in decline, they still play an important role in the landscape for soil protection and act as barriers and boundaries between management units (Baudry et al. 2000). The review concluded that an examination of the distribution of hedgerows would aid in the design of appropriate policies for landscape management and conservation of hedgerows. The study also detailed the important role hedgerows play in the ecological health of the countryside and the various social, historical, ecological and production functions as introduced above (Baudry et al. 2000).

Buffer strips

A major focus of the review by Wenger (1999) was on the impacts of riparian buffers on phosphorus, nitrogen, other contaminants and other factors influencing aquatic habitat.

Perennial vegetation on buffer strips also has a key role in nitrogen-buffering capacities, since it retains nitrogen through uptake during the growth season which corresponds to the period of low water tables when denitrification is limited by soil aeration (Vought et al. 1995). Riparian vegetation acts as a mechanical filter for suspended matter and sediments during floods, and can reduce the pesticides and herbicides input to the stream. The benefits vary according to the nutrient load, hydraulic conditions and the type of vegetation cover (Vought et al. 1995).

Costs

The Environment Agency commissioned work on the long-term costing tool for flood and coastal management (Environment Agency 2015a). It provides evidence in a collection of reports and offers a spreadsheet tool that can help practitioners build up whole life cost estimates.²⁰ The spreadsheet of relevance is the 'Cost estimation for land use and run-off -

²⁰ Project SC080039. See <u>http://evidence.environment-agency.gov.uk/fcerm/en/Default/FCRM.aspx</u>

summary of evidence'. The section on 'Soil management' includes minimum tillage, buffer strips and conversion of arable to extensive grazing.

Defra's Mitigation Options for Phosphorus and Sediment (MOPS) project investigated the efficacy and cost-effectiveness of different in-field mitigation options to control diffuse pollution losses in surface run-off from arable fields. Cost estimates from the MOPS project for minimum tillage concluded that minimum tillage should not increase costs. In-field barriers (or buffer strips) of a 2m wide vegetative barrier on the contour included a one-off capital cost for the establishment of £3–5 per 100m for a 2m width barrier, and an ongoing annual maintenance cost for topping the vegetation £0.5–0.6 per 100m of a 2m width barrier (Environment Agency 2015c). Environment Agency (2105c) also outlines the detailed costings of a riparian buffer strip from the estimations from the Water Framework Directive (Scottish Government) and the Collaborative Research programme on River Basin Management Planning Economics.

4.2.5 Effectiveness/performance

This literature review has determined how different soil and land management measures can increase soil storage and increase the water flow path travel time. However, the literature reviewed does not make the final step towards determining their impact on reducing flood risk. The effectiveness or performance of soil and land management measures on flood risk management are often implied, rather than modelled or measured as run-off rates, peaks, storage, routing effects, total run-off or number of households protected. The modelling of a number of these land management measures (for example, conservation tillage, cover crops, buffer strips and hedges) is often done by increasing the Manning's n hydraulic roughness parameter, although other model parameters representing 'losses', such as inception, evaporation and infiltration can also be modified in some models. However, there is still uncertainty over the accuracy of Manning's n value and thus the use of empirical data measured by the roughness or flow velocities for different land cover types is still needed. This gap in evidence can be explained by the complexities described in Section 4.3 to determine how a change in soil and land management can reduce flood risk.

Although there is limited evidence that describes how soil and land management reduce flood risk management or total run-off, there are often measurements of soil porosity, soil moisture content, infiltration capacity and other variables. For example, infiltration capacity can be predicted from measured soil hydraulic properties and an initial moisture state. By measuring infiltration rates under ponded (positive soil water pressure) and matric tension conditions and conducting a comparison, it is possible to infer macropore hydraulic conductivity and porosity (Strudley et al. 2008). While this does not necessarily imply the presence of a flood risk management function, it can modify run-off, flood flows or flood peaks, in which more information would be needed to determine if it impacts flood risk. These measurements of flood risk management are examined below.

Soil aeration and subsoiling

The success of subsoiling is dependent on the complex relationship between the clay percentage, mineralogy and pH of the soil. It is also dependent on the soil moisture conditions during the operation and the depth of the operation (Castle et al. 1984).

Smith (2012) conducted a rainfall simulation experiment on the effect of soil aeration on field hydrological properties. The results showed that, in lightly compacted fields, the effects of soil aeration can vary significantly from negligible effects to increasing soil water storage capacity by up to 100%, and can delay run-off peaks. One simulation showed the effects of soil aeration routing the water down through the surface soil layer, allowing a majority of the rainwater to be conveyed across the field as shallow throughflow (Smith 2012). In heavily compacted fields, soil aeration did very little to improve the fast run-off response to rainfall

because the compacted layer of the soil was so impermeable and deep that the slits caused by the aerator made no difference to the routing of the rainwater (Smith 2012). It has been suggested that several runs of the soil aerator in different directions could improve the response in heavily compacted fields.²¹

Arable systems

Conservation tillage

The recorded effects of conservation tillage on run-off are variable and depend on a number of soil, land and climatic factors. Hill and Mannering (2004) determined that:

- a 30% residue cover from a conservation tillage system reduced soil erosion rates by 50–60% compared with conventional tillage
- the plant residues that remain improve soil structure and increase soil water infiltration and soil water storage capacity

Reduced tillage has been shown to present benefits over no tillage, although they are less substantial (BIO Intelligence Service and HydroLogic 2014). In Spain, run-off from reduced tilled plots represented about 80% of run-off from the non-tilled soil (BIO Intelligence Service and HydroLogic 2014). Reduced tillage was also slightly more effective in rainfall capture, and allowed a reduction in the annual irrigation water need of the tree plantations of about 6–9% (Abrisqueta et al. 2007).

Despite the advantages of conservation tillage for soil water retention, environmental tradeoffs may include increased use of herbicides, which is often associated with these practices for weed contro. However, the use of pesticides could be reduced through appropriate rotations and use of cover crops (Melander et al. 2013).

Early sowing and cover crops

The effect of increasing crop cover to reduce run-off was reported by Davies et al (1972). However, the success of cover crops and soil mulches on reducing field plot run-off appears to be more uncertain and dependent on soil type (O'Connell et al. 2004). In practice, the results vary from 80% reduction in surface run-off using winter cover crops in Germany (Schafer 1986) to no significant difference using under-sown rye grass or winter cover crops and the UK (Environment Agency 2002a). Cereal rye is an excellent winter cover crop because it rapidly produces a ground cover that holds soil in place against the forces of wind and water while requiring limited amount of water (BIO Intelligence Service and HydroLogic 2014).

Environment Agency (2008a) states that crop covers can reduce the land at risk of run-off and erosion on sandy and light silty soils when winter cereals are sown during late October and November, reducing the risk of soil surface becoming capped. Or, the early sowing of winter crops can increase the level of vegetation cover which will slow run-off over the winter months.

(2009a) suggested that, where any harvesting takes place or if forage crops (for example, kale and stubble turnips) are grazed in winter or under wet conditions, primary cultivation should be undertaken as soon as conditions are suitable to create a rough surface that will reduce the risk of run-off and erosion. To minimise run-off and erosion before spring-sown crops, it was recommended that a temporary green cover is established or to leave the land in stubble or roughly cultivated over winter (Defra 2009).

²¹ Personal communication by T. Dawson (2011) to K.A. Smith regarding the use of a soil aerator on heavily compacted soils.

Defra (2005) recommended that the sowing of winter cereals should be early enough to ensure 25% cover by early winter, that is, no later than mid to late September in lowland England. For spring cereals, Defra (2005) does not recommend early sowing, particularly on fields with high erosion risk.

Crop rotations

There is no known literature that determines the effectiveness or performance of crop rotations in reducing flood risk management. A report for the European Commission does note that the introduction of legumes (such as beans and peas) or ryegrass – even in short rotations – may enhance the soil water retention capacity through limited water consumption, appropriate soil cover and structural properties (BIO Intelligence Service and HydroLogic 2014).

Grassland systems

Bilotta et al. (2007) conducted a comprehensive review of literature relating to the impacts of grazing animals on the quality of soils, vegetation and surface waters. The research showed that the impact of grazing animals on soil hydraulic conductivity is dependent on the amount of pressure exerted on the soil, which is in turn dependent on the species and age of the grazing animal. In turn, the amount and form of soil structural alteration is a result of this force determined by the stocking density, soil moisture content, soil texture, and the presence/absence of a protective vegetation cover (Bilotta et al. 2007). Consequently, these factors are a part of determining the effectiveness and performance of changing the regime of grazing animals on grassland soils to increase soil infiltration rates and soil water retention capacity, and consequently reduce flood risk.

The review by Gifford and Hawkins (1978) on experimental studies concluded that, as a rule of thumb, light grazing reduces infiltration by a factor of 0.25 and heavy grazing by a factor of 0.50. The study by Heathwaite et al. (1990) determined that the infiltration capacity on grazed land could reduce by 80% if stock trampled and compacted the soil surface. They found that:

'run-off from heavily grazed permanent grassland is at least double that from lightly grazed areas, and nearly twelve times greater than that of ungrazed (temporary grassland) area'

Reducing the intensity of trampling or use of slot cutting or a spiking machine can reduce the compaction of the soil, allowing an increase in surface infiltration, and improving structural condition to increase soil storage (Environment Agency 2003b). The removal of livestock generally leads to a reduction in surface flow volumes, which is attributed to an improvement in the structure of the upper layers of the soil, enhanced infiltration and evaporation (see, for example, Gifford and Hawkins 1978, Greenwood et al. 1997, Nguyen et al. 1998, Greenwood and McKenzie 2001 and Carroll et al. 2004b).

Another factor in livestock and grazing management is the type of vegetation present or planted in the grassland system. Macleod et al. (2013) examined the hydrological effectiveness of the grass hybrid species *Festulolium Ioliaceum* cv Prior compared with the grass species of choice for most farmers (perennial ryegrass, *Lolium perenne*) and a more stress-resistant grass species, meadow fescue (*Festuca pratensis*). An experimental design was conducted on a planar hillslope (5°) on a clay-rich soil and subsequently maintained over 3 entire growing seasons (2006 to 2009) at Rothamsted's North Wyke Research Station in Devon. The hybrid species, Prior, showed that it reduced run-off during the events by 51% compared with *L. perenne* and by 43% compared with *F. pratensis* over a 2 -year field experiment (Macleod et al. 2013). It was concluded that this reduction in run-off could be attributed to the intense initial root growth followed by rapid senescence, especially at depth. The senescencing roots of the Prior that penetrated the deeper soil layers were likely

to leave bio-pores that can transmit water to depth, resulting in a greater soil water storage capacity (Macleod et al. 2013).

The Culm grassland study is another important study. This measured and monitored the water retention capacity of the Culm grassland in Devon in relation to that of intensively managed grassland, and scrub and woodland (Puttock and Brazier 2014). The results suggested that Culm soils (unimproved Culm grasslands) stored more water than intensively managed grasslands, and scrub and woodland. The water stored in the soils varied over time, but mean estimates for Culm grassland (~241 litres per m² surface area) were significantly higher than intensively managed grasslands (~62 litres per m⁻² surface area) (Puttock and Brazier 2014). Although this study found a significant increase in the water retention capacity of the Culm grassland in relation to intensively managed grasslands, the study site was within a unique ecological area known as the Culm National Character Area and the findings may not be applicable to other landscapes. The Culm National Character Area covers 3,500km² in south-west England and is an area of international conservation importance which includes wet, unimproved, species-rich pastures typical of poorly drained acid soils, supporting a suite of purple moor-grass and soft rush communities (Puttock and Brazier 2014).

Agricultural landscape features

Hedges

A number of factors determine the effectiveness and performance of hedges on their environmental, ecological and cultural functions including the size of hedges, their ecological and structural diversity, management and location, and the wider network of hedges in the landscape (Baudry et al. 2000); these factors would also be applicable to flood risk. Although hedgerows have been used in land management for decades across the world to control water (Baudry et al. 2000), no evidence was found on the effectiveness or performance of hedgerows in managing flood risk.

Buffer strips

There is limited evidence to demonstrate the effectiveness and performance of the hydrological function of buffer strips in relation to run-off reduction, both at the plot and catchment scale (Lane et al. 2007). Borin et al. (2010) completed a number of experiments in Italy to further investigate the impacts of buffer strips on reducing agricultural diffuse pollution in lowland areas, and consequently measured the total run-off. The main experiment showed that a 6m wide buffer strip (2 rows of regularly alternating trees, *Platanus hybrida* Brot. and shrubs, *Viburnum opulus* L.) at the Podova University Experimental Farm reduced run-off by 78% compared with no buffer strip between 1998 and 2002, equivalent to a run-off depth of 231mm over 5 years. The ancillary Mogliano experiment (one line of trees and a strip of grass for a total width of 4m) measured total run-off from 2003 to 2005, finding that the total run-off without buffer strips was about 97mm and 61mm with buffer strips (Borin et al. 2010). Since the aims of the experiments were centred on reducing agricultural diffuse pollution in lowland areas, there was little interpretation or analysis of the total run-off measurements.

Gao et al. (2016) conducted a modelling study of land cover change impacts on flood peaks in 3 upland peat basins. The study concluded that a wider strip with higher density vegetation (for example, Sphagnum) leads to a lower flow peak and delays the peak. A narrower buffer strip on the hillslopes surrounding upstream and downstream channels had a greater effect than a thicker buffer strip just based around the downstream river network.

A cautionary note was summarised by Stutter et al. (2012) who noted that, increasingly, buffers are expected to act as 'end of pipe solutions' to the terrestrial environment's outputs

at the expense of appropriate source controls. As such they may become overloaded during larger events.

4.2.6 Key case studies

There are limited UK case studies that focus specifically on the topic of land and soil management with quantifiable evidence of their direct impact on flood risk at the catchment scale. Most of the studies focus on biodiversity and pollution. There is some effort needed to tease out both hydrological and catchment flood impact evidence.

The **Loddington Case Study (Leicestershire)** is key to determining the impact of minimum tillage, contour cultivation and no tramlines in fields on event total run-off responses to rainfall, peak run-off and the timing of the run-off peaks in order to assess whether these mitigation options could mitigate downstream flood risk (Deasy et al. 2014). A mixed modelling approach was adopted to determine whether differences observed in run-off responses were significant. The results from this study suggested that an impact on local scale run-off generation, with differences observed in the size, duration and timing of flood peaks with changes in land use management (Deasy et al. 2014).

The **Pontbren Case Study** is often cited to discuss woodland management as a form of WWNP project. However, this case study also has important lessons for understanding community engagement and conflicting results of the hydrological processes in ungrazed versus grazed land, and other forms of soil and land management such as hedges (Marshall et al. 2014). In the Pontbren catchment (18km²), the farmers decided to (Posthumus and Morris 2010):

- reduce stock numbers (using de-stocking grants)
- switch to hardier sheep breeds that stayed outdoors all year round
- plant trees and hedges in order to provide shelter

As they started to see environmental benefits, they decided to carry out more work, such as the creation of ponds and fencing off water courses (Posthumus and Morris 2010). Others soon became interested. This led to the formation in 2000 of the Pontbren Group, an informal co-operative which is still active today. The results from the study by Marshall et al. (2014) did not show significant differences between grazed and ungrazed land in soil infiltration rates and soil physical properties. Further information can be found in Section 4.2.3 and the Marshall et al. (2014) paper.

The **Dartmoor Culm grass study** also produced encouraging outcomes in many respects. It shows how multiple benefits and a new way of valuing the land can drive land use change. The lower intensity farming and the fundamental changes to basic hydrologic parameters such as soil storage, surface infiltration and roughness all point to potential food risk benefits, even if it is difficult to quantify at a larger scale (Puttock and Brazier 2014). It suggests a landscape with a mix of higher and lower intensity farming methods to give an overall lower flood risk at the catchment scale (Puttock and Brazier 2014). More information can be found in Section 4.2.5.

4.2.7 Design, management and maintenance

The evidence shows that the design, maintenance, and management requirements of soil and land management measures to reduce flood risk depends on multiple options that can be predetermined (that is, climate factors, soil type) and governed by human intervention (that is, equipment used, type of cultivations).
The first step in determining the design, maintenance and management requirements is to carry out a soil structure assessment. The field-based method of soil structure assessment is relatively quick and cheap to carry out with each observation taking between 10 and 15 minutes, and a field diagnosis of structural damage and any necessary remedial actions can be identified in less than an hour (Palmer and Smith 2013). It is important that farmers and advisers become proficient in identifying soil structural condition and that assessments become part of the routine soil management on the farm (Palmer and Smith 2013). By completing this assessment, farmers and others will be able to better identify the optimum soil and land management measures that can address flood risk management and achieve other benefits.

Soil aeration and subsoiling

There are 2 main types of subsoiling: convention and winged subsoilers. Other types include those that loosen the soil with minimal surface disturbance at shallow lengths, or a multi-tined implement with a vibrating tool bar also intended for shallow subsoiling (Castle et al. 1984).

Research has shown that there is a 'critical' depth, below which soil loosening does not take place but where the soil is displaced sideways and a channel is formed (Castle et al. 1984). There is no specific critical depth for any type of soil; it depends on the soil's physical condition and may range from close to the surface in a wet soil to a depth of 450mm in drier soils (Castle et al. 1984). The critical depth can be modified by loosening the surface layers first or increasing the effective working width of the subsoiler foot (Castle et al. 1984).

There is an optimum depth in which the subsoiling should be conducted when dealing with compaction, but this can only be determined by trial and error. It is important to dig a profile pit to examine the subsoil before subsoiling; however, the depth of this compaction can vary significantly across the field and therefore it is essential after commencing the operation to dig down and check that the problem area is being disturbed as required (Castle et al. 1984).

The spacing of subsoiling is important to the extent in which it loosened the soil. It is not absolutely essential for a uniform disturbance to deal with water ponding in the upper layers of the soil; however, it is important for root development and penetration (Castle et al. 1984). If there is not uniform disturbance, it may increase risk in grasslands or where direct drilling is carried out.

Subsoiling should only occur during optimum soil moisture conditions, when the subsoil is dry though not excessively so (Castle et al. 1984). There are no current guidelines on the optimum soil moisture content, so it is recommended that a number of trial runs occur. In high rainfall areas, soil moisture conditions occur less frequently, although surface conditions can be firm (Castle et al. 1984).

Lastly, subsoiling should not occur too frequently to reduce the risk of further compaction. The optimum frequency of subsoiling is dependent on the soil type, the nature of compaction, climate and management (Castle et al. 1984).

Arable systems

There are many different ways to design, maintain and manage arable systems. These rely on a number of different factors that have been discussed throughout this literature review including soil and climate characteristics, crop yields and production.

The Code of Good Agricultural Practice (CoGAP) outlines objectives for soil and land conditions (Defra 2009). The design, maintenance and management of conservation tillage, early sowing and cover crops, and crop rotations will be designed to achieve these

objectives. In general, it states that well-drained and well-structured soils allow water to enter more quickly and therefore should reduce the risk of run-off and erosion. See Defra (2009a) for further details.

CoGAP recommends sowing a temporary cover or catch crop on suitable soils in early autumn when an early harvested crop is to be followed by a spring-sown crop since this cover crop will take up nitrogen and reduce leaching losses. Additionally, winter green cover is particularly important on a one-year set aside land (land that is not being cultivated). Autumn-sown crops should be sown as early as possible to reduce the risk from pests and diseases. Crops sown in early September will take up more nitrogen than later sown crops, and will also reduce the risk of run-off and soil erosion. It is also recommended that the residues of late harvested crops, such as root crops, are left undisturbed until the following spring unless the soil is compacted and there is risk of run-off or soil erosion (Defra 2009).

These recommendations highlight the need to monitor the structure of soil regularly, creating a soil management plan to determine the next actions or inactions for the best soil structure and properties for reducing flood risk or other benefits. For example, inspecting a spade full of soil can help identify the soil structure in each field (or part field) to determine how to maintain or improve its condition (Defra, 2009a). Recording the steps taken on a field-to-field basis on how run-off and erosion were minimised to ensure food structure and maintain the infiltration of rainfall will make up part of the management plan. Other circumstances that cause ponding, run-off or erosion should also be recorded to determine how the soil reacts to rainfall events and determine how a soil and land management plan must be conducted (Defra 2009).

Grassland systems

Similar to arable systems, the design, maintenance and management requirements of grassland system measures depend on a large number of factors. CoGAP is the guidance to help farmers, landowners and others to select the appropriate actions for each individual situation. CoGAP outlines good practice for managing livestock in a way that protects grasslands and soils to maintain productivity, and reduce the risk of causing damage to the surrounding environment (Defra 2009). Minimising the impact of stock and its related management operations on the soil will reduce the risk of run-off and erosion. By constantly monitoring the structure soil, those managing the land will be able to determine when stock should be removed from the land when the soil is too wet and poaching becomes a risk to subsequent production and run-off or erosion will pollute surface waters (Defra 2009).

Agricultural landscape features

Hedges

Hedgerows are mostly observed as land boundaries, but can also act as natural breaks to overland flow. However, the siting of hedgerows is rarely determined to minimise flood risk, as the visual effects of the historical landscape and wildlife must also be considered (Defra 2009).

This literature review did not find any peer-reviewed literature that determined the design, maintenance and management of hedgerows particularly for flood risk. There are a number of best practice guides to hedgerow management produced by local authorities and non-governmental organisations, but these do not explicitly tailor guidance for flood risk management. Examples include:

- 'Countryside Hedgerows: Protection and Management' (Natural England 2016a)
- 'Farm Hedges and their Management' (RSPB 2008a)

- 'Farming for Wildlife in Wales: Hedgerow Management' (RSPB 2006)
- 'The Complete Hedge Good Management Guide' (Hedgelink UK 2013)

Buffer strips

CoGAP suggests that buffer strips be planted alongside surface waters which are at the bottom of the slopes. It is, however, noted that buffers alongside surface water may be less effective in river catchments where the majority of water flows below the land surface (Defra 2009).

Buffer strips are commonly situated in the riparian zone and they are usually of a limited width. A buffer zone can occur anywhere in the catchment and should be sited to intercept pathways of concentrated surface run-off (Lane et al. 2007).

In a recent Danish evaluation of buffers (Kronvang et al. 2010, 2011), it was concluded that the mandatory 10m wide riparian buffers would have been more cost-effective if buffers were targeted at critical areas. However, targeting buffer strips of variable widths according to the risk of pollution to watercourses requires a regulatory approach that is flexible and that can provide for the inherent costs in guidance, assessment of compliance, and administration of compensation payments (Stutter et al. 2012).

Qui (2003) proposed a strategy for placing conservation buffers based on the variable source area (VSA) hydrology. VSAs are small, variable but predictable portions of a watershed that regularly contribute to saturation-excess overland flow generation. This VSA-based strategy involves 3 steps (Figure 4.13):

- Identify VSAs in landscapes based on natural characteristics such as hydrology, land use/cover, topography and soils.
- Target areas within VSAs for conservation buffers.
- Refine the size and location of conversation buffers based on other factors such as weather, environmental objectives, available funding and other best management practices.





Source: Qui (2003)

This strategy increases the relationship between the hydrology of the agricultural landscape and conservation buffers (Qui 2003). However, there is no user-friendly and well-documented biophysical simulation model to identify the VSAs (Qui 2003). A simple

approach, however, can be provided by mapping the topographic index or by utilising the knowledge of the farmer or landowner to determine the areas of saturation during storm events to determine the siting of buffer strips.

4.2.8 Research gaps

The main gap identified by this literature review is that there is no knowledge of the impact on flood risk of the soil and land management measures reviewed. The evidence has determined how different soil and land management measures can increase soil water retention capacity, soil organic matter and travel time. However, the literature reviewed does not make the final but crucial step towards determining the impact on reducing flood flows, peaks and lag time.

The modelling and prediction of the hydrological impacts of land use change continue to remain an evidence gap (Wheater 2002, O'Connell et al. 2007). To determine the effect of soil and land use management measures on flood risk, further data and assessments are required to compare against other WWNP measures and traditional flood risk management measures. Although there is limited evidence that these measures increase soil hydraulic properties on a local scale, this further assessment would also guide practitioners in determining if upscaling soil and land management would have tangible benefits at a catchment scale.

Currently, there is a multiscale modelling problem of upscaling to represent impacts at catchment scale. While new methods have recently been developed to represent the effects of changing soil properties and vegetation for upland land management (Jackson et al. 2008, Wheater and Evans 2009 and Wheater et al. 2008), they have been supported only by a single extensive experimental data set (Wheater and Evans 2009).

Current research is intended to generalise these modelling tools for a wide range of land use issues, but the results will be subject to high uncertainty without more extensive data (Wheater and Evans 2009). Further research needs to be conducted to clarify how increasing measures on a larger scale affect other complex tributary interactions and flood risk downstream (Lane 2017). The impact of a local NFM scheme needs to be tested in larger scale river networks to address any synchronisation issues (see Section 4.1.6). Practitioners need better guidance and tools to assess synchronisation issues and in the deployment of spatially variable NFM schemes in multiple subcatchments. By addressing these synchronisation issues, the antecedent conditions will be explored, something which also identifies a gap in the reliability and performance of these interventions varying between rainfall events across catchments (Lane 2017).

More specifically, while the capacity of the soil to intercept and hold water and the function of the available soil storage are known, the natural variability of soils makes upscaling difficult. The role of soil in storing water, suppressing flow and slowing flow are well-established, but the performance of soil management techniques during large events is the key research gap. This gap includes determining the impact and performance of the soil under different antecedent conditions.

Rogger et al. (2017) also identified the need to better understand the dynamic nature of soil structure and its effects on hydrology, particularly how the seasonal variations of soil hydraulic properties are modified by tillage, compaction, cracking by repeated shrinking and swelling, and soil sealing processes. Furthermore, the effects of seasonal variation at scales larger than the plot scale need to be better understood. They concluded that a better mechanistic description of coupled mechanical and hydraulic processes is needed to capture the evolution of soil structure by tensile forces and compressive loads, and biological effects on soil structural characteristics, such as preferential flow pathways through macropores induced by earthworms and root penetration, and others (Rogger et al. 2017).

A gap has also been identified in conducting long-term field experiments that encourage systems thinking that includes spatial and temporal dimensional factors, and focus on connectivity and spatial patterns (Rogger et al. 2017). Additionally, a meta-analysis of reported studies would assist in developing a common framework for organising results, including developing a criterion of similarity, to help increase the applicability of particular catchment studies (Rogger et al. 2017).

The multiple benefits generated by modified soil and land management measures still need to be better quantified in the overall cost–benefit analysis of an NFM scheme. Arguments can be made in 'valuing' our landscape more effectively and setting new visions for future land, food and water management (Ozdemiroglu and Hails 2016). The examples from the Devon Wildlife Trust and Culm grassland were very compelling in using this approach based on wider ecosystem benefits (North Devon Wildlife Trust 2014).

Additionally, further research is needed on the interactions between climate impacts and soil and land management, including uncertainties in socioeconomics and the greenhouse gas emissions scenarios derived from them and the changes in future climate their impacts as a result (Falloon and Betts 2010). Some climate impacts are often affected by socioeconomic and land use changes, so there needs to be a consistent application of socioeconomic, land use, emissions and climate data to impact assessments with integration of water cycle–ecosystem–climate models with socioeconomic simulations (Falloon and Betts 2010).

There is evidence and guidance to promote good soil and land use management practices, and how these could be optimised in different agricultural systems. There may be a need for a new breed of models to capture system level changes and express these in a suitable riskbased format to encourage and fund the uptake of good soil and land use management.

4.3 Land and headwater drainage management

4.3.1 Introduction

This section looks at the overall drainage management of catchment headwaters and lowland areas, and how this affects flooding. For the purposes of this section, headwater catchments are loosely defined as typically small catchment areas up to several square kilometres in size in river source areas containing both intermittent streams and perennial streams.

Within these headwater drainage networks, there are potential opportunities to intervene to change the storage and the travel time of water within them, thereby modifying the delivery of quickflow further down the arterial drainage network. Some studies of headwater catchments have focused primarily on flow regime, while other studies include elements of flow regime analysis but may have been more focused on other issues including water quality, erosion and habitat elements of catchment management.

This section reviews a number of these more widely focused studies, along with studies that are focused on headwater discharge, to identify any evidence that illustrates that changes to headwater management will yield run-off change and flood risk reduction. This section does not cover the flow pathways through soil infiltration, soil water flow and underdrainage. Further information on these topics can be found in Section 4.2. This section also does not include larger structures such swales, bunds and ponds as these are covered in Section 4.4.

This section concentrates on NFM interventions within 2 specific location types:

- at a field scale and along tracks and farm roads
- within drain/ditch systems on farmland and peatland systems

The section discusses a range of different techniques with the aim of assessing how individual techniques on a local scale can affect the delivery of quickflow and their aggregated impact at a catchment scale. A short description is given of the unique nature and characteristics of lowland drainage systems and the associated complex management of water levels and flood risk (for example, using pumps and sluices) to deliver a range of ecosystem services. However, to date, there have been very few studies on the application of an NFM approach specifically within lowland basins.

4.3.2 Understanding the science

This section assesses field and track scale overland flow interventions and ditch/drainage interventions in 3 different settings:

- agricultural headwater drainage
- headwater peatland restoration
- lowland water level management

Agricultural headwater drainage

Surface water flow can contribute to flooding and, when not controlled, can lead to a variety of damaging effects on agricultural headwaters. This section considers 3 flow pathways:

- through fields
- on tracks, paths and roads on farms
- through man-made ditches

Control measure options to reduce and change surface flow pathways in each route of surface flow in order to reduce flood risk are discussed.

Flow pathways within fields

Tramlines associated with arable farming and wheelings across fields can intercept surface flows and concentrate flow into paths along their length depending on the flows (Schwab et al. 1993, Environment Agency 2008a). The problem of localised compaction from farming equipment exacerbates the issue of intercepting surface flows Figure 4.14).



Figure 4.14 Headlands and wheelings causing run-off (left) and attempts to use cross-slope cultivation to disrupt long flow pathways (right)

Source: Environment Agency (2008a)

Withers et al. (2006) documented the nature of this overland flow problem and the potential to intercept these flow pathways. The first line of defence to reduce the concentration of flow from entering tramlines associated with arable farming and wheelings, and consequently to reduce flow risk, would be to reduce the compaction within the tramlines (Withers et al. 2006). There are some remediation techniques when creating the tramlines to reduce compaction. Tillage can act as an interception measure by breaking compacted surfaces and so can help disrupt the flow (Environment Agency 2008a). Additionally, reducing the density of tramlines by increasing the width of the boom also reduces the concentration of surface flow (Withers et al. 2008). Figure 4.14 illustrates the use of cross-slope cultivation to disrupt long flow pathways.

Flow pathways from tracks, paths and roads on farms

Tracks, paths and roads on farms can concentrate flow along their length due to their more impermeable nature and smoother bed. Tracks will either be artificially constructed for access purposes, but some will have been created by animals. Sheep often follow the same routes across fields, thereby creating more compacted linear tracks which increase hydrological connectivity (Zhao 2009).

Track or path interception, such as hump cross drains or channel cross drains (Figure 4.15) which can reduce the concentration of flow along pathways, will reduce connectivity and have the potential to reduce flood risk, although it still remains to be tested. Such an approach has not been adopted for animal tracks, and largely focuses on 'made roads'. The evidence behind these track or path interception techniques is mostly based on research looking at the connectivity of flow for sediment and phosphorus losses. However, Collins (2012), Edwards and Withers (2008), and Srinivasan and McDowell (2009) show that these flow pathways on farms play a role in flow connectivity. Evans and Boardman (2003) reported that managing track run-off connectivity in the South Downs, West Sussex, is effective for 'muddy' flood (surface water sediment-laden flood) control. Withers et al. (2006) further showed that flow on tramlines can end up on roads and farm tracks, often by passing through field-access gateways. Evans (2006) showed that track management that diverted surface flow into ponds helped resolve many of the local 'muddy flood' issues, but management of the ponds was very important due to sedimentation. Further down the flow path network, the topographic regime takes over and the larger flow pathways follow the path of the valley floor (Chambers et al. 1992).



Figure 4.15 Operation of hump cross and channel cross drains

Source: FWAG South West (2016)

Cross drains (flow capture features that cross tracks; Figure 4.15) or any track or road-based drainage feature can divert flow laterally onto fields or into ponds, and have the potential to increase the travel time of overland flow. Two examples of track-based interception are shown in Figure 4.16. The left-hand image in Figure 4.16 is taken from a study at Nafferton Farm (Quinn et al. 2008a), where a known fast and polluting flow pathway was connecting run-off from farm hardstandings to the main ditch. To control the polluting flow pathway, a standard manhole was placed on the track and a pipe redirected the flow from the road under the hedge and into field pond close to the main ditch. However, the drain did need maintenance if all the flow was to be captured by the drain. The right-hand image in Figure 4.16 illustrates the Eden DTC study where a number of on-farm measures had been installed primarily for diffuse pollution management (Eden DTC 2017). It shows a cross drain placed across a known active surface pathway. This is 1 of 6 cross drains placed along a 175m of track that traversed the farm. The water then flows through a grass field and into a sediment trap and ditches (an example of which can be seen in Section 4.4), which act together to retard flow. All the features aimed to increase travel time by breaking up the connectivity of the surface drainage network. Fast flow pathways have been targeted by local knowledge and experience, but it is difficult to quantify the impact at the headwater scale for flood reduction.





Figure 4.16 A known flow pathway on a track is drained and piped to a nearby field area (left) and a cross drain directing surface flow onto a grass field (right)

Source: Quinn et al.(2008a), Eden DTC (2017)

This section focuses on tracks in a farm setting. Experience from run-off on tracks in a large forest/woodland setting can provide complementary knowledge (see Chapter 3).

Flow pathways within ditches

Open drainage ditches have the advantage of large carrying capacities and for this reason they have always been an important feature of agricultural land drainage systems (Brady 1974). There are various types of ditch, with some used as outlets for underdrainage to transport the water away from land to a watercourse, some to intercept surface flows and promote infiltration, and some to help regulate water table levels locally. This section focuses on ditches created for drainage purposes rather than those that perform other functions such as the distribution of water for irrigation.

There are options to change the form and function of the ditch, either small or large, which can influence the travel time of water through them such as:

- widening the ditch
- increasing hydraulic roughness by the choice of in-channel vegetation or rougher bed and bank materials, and its management
- partially blocking the ditch

• online sediment traps

Widening the ditch

Shore et al. (2015) investigated the relationship between ditch size and function. They stressed that there is no design criteria for the size of the ditch, and often the ditch is oversized for the area of land it is meant to drain. The perceived agronomic wisdom is that a well-maintained ditch is a good ditch (Shore et al 2015), fulfilling its primary function of removing water quickly and optimising the drainage function of in-field underdrains. Widening a ditch (see Figure 4.17) in particular areas of the ditch system increases the travel time, and these widened ditch areas are often used as a sediment trap. Online sediment traps, which do not include widening the ditch, are discussed further below.



Figure 4.17 Conversion of a conventional ditch to a two-stage ditch

Source: Kobell and Journal (2015)

Increasing hydraulic roughness

There are a number of studies that conclude qualitatively that increasing hydraulic roughness (such as vegetation) in ditches can reduce flood flow. Whitworth (2011) and The Rivers Trust (2014) concluded that the reintroduction of vegetation can slow and clean the flow. Vegetation can also play a role in increasing travel time and evapotranspiration when widening ditches. Figure 4.18 and Figure 4.19 are examples of ditch management using vegetation within the channel.

In-channel vegetation acts as a filter between flow leaving the field and the ditch; the ditch can still perform its primary function of removing flow from the field area (overland flow and/or underdrainage flow) but the flow in the ditch could be slower. Overall, in non-flood conditions, the vegetated drain function of draining the local water table is not impaired, except for when the vegetation is so dense that low flow water levels are raised in the ditch which could eventually compromise the local field underdrainage function (Eden DTC 2017). Physical reasoning when applying hydraulic equations (for example, involving Manning's n) suggests that a positive outcome for flood management will be achieved when roughness is increased. However, the actual (quantifiable) impact of the features is difficult to estimate at the catchment scale.

Figure 4.19 (left) shows a zone of willow planted in the ditch bed; this is 1 of 3 zones (10m in length), with each zone having a separate variety of willow. All zones show the ability to back up and attenuate flow, including during the winter period, when most of the other ditch vegetation had died back. Figure 4.19 (right) shows a single willow barrier, a so called 'living barrier' as the small shoots will eventually grow into quite large trees (up to 5m high). There was no attempt to create a specified spacing or any ideal habitat for the feature. The 'living barrier' is 10 years old and the farmer has had no problem with the willow in the ditch. This trial has resulted in a desired outcome of an all year round attenuation of flow. It is, however, very difficult to quantify the peak flow reduction as the leakiness is highly nonlinear in nature and changes in time. The study included 10 interventions in the ditch. Observations of flow

were taken upstream and downstream of the features some 400m apart. The results showed the measures had differing degrees of storage and attenuation effects dependent on antecedent conditions and the storm magnitude (Quinn et al. 2008a). It has proved very difficult to disentangle and detect the changes occurring due to the 10 features. As the flood wave was routed along the ditch (over the 400m), it attenuated the flood flow which lowered the flood peak. Equally, the flow was also known to be increasing along the length of the ditch as more lateral subsurface flow and drain flow entered the ditch.





Figure 4.18 Ditch vegetation attenuating flow (left) and a vegetated ditch (right)

Source: Whitworth (2011), The Rivers Trust (2014)





Figure 4.19 Willow growth within a ditch (left) and living barriers (right) will establish a permanent self-maintained leaky structure

Source: Quinn et al. (2008a), Quinn (2016b)

Partial blockage and online sediment trap

Ditches can also be partially blocked or altered, and in turn alter the run-off function in a number of ways. Figure 4.20 (left) shows part of the Eden DTC, where water flowing both from a land drain and from overland flow in a field is slowed and backed up by a small stone barrier (which is one of 5 over a 100m reach length). On a larger scale in Figure 4.20 (right), a ditch remeandering exercise in Selkirk lengthened and roughened a flow pathway that was originally steep and fast-flowing (SEPA 2015). Hydraulic models could be parameterised in

the future to represent these features in order to estimate a change in discharge at different scales.



Figure 4.20 A partially blocked small drain in the Eden DTC (left) and a meandering a small ditch in Selkirk (right)

Source: Eden DTC (2017), SEPA (2015)

Figure 4.21 shows 2 example trials of within-ditch features at Nafferton Farm (Environment Agency 2012, Flow Partnership 2017). In the Nafferton Farm project (and as reported in Jonczyk et al. 2008), a number of within-ditch features were built and tested. The feature in Figure 4.21 (left) targeted sediment and phosphorus reduction. To maintain the function of the feature during larger storm events, the barrier was designed to attenuate flow (see Section 4.3.5 for more details). It was difficult to estimate the flow travelling through the barrier and the residence time. When full, the water would flow over the feature, protecting the surrounding fields from flooding, until bankfull height. Figure 4.21(right) is a similar feature on a small ditch constructed as part of the MOPS project (Ockenden et al. 2012). The team constructed a number of ditch features, which they referred to as 'in-line wetlands' in modified ditches. The 10 features constructed all showed a good reduction in phosphorus and sediment levels. It was difficult to determine the flow reduction or residence times. Ockenden et al. (2012) suggested that there was some potential flood attenuation benefit along with biodiversity and pollution reduction.



Figure 4.21 Widening a ditch and adding a recycled leaky plastic barrier at Nafferton Farm (left) and a widened ditch intercepting a known overland flow pathway (part of the MOPS project) (right)

Source: Quinn et al. (2008a), Ockenden et al. (2012)

As part of the Cheviot Futures project, Barber (2013) provided the design of an online sediment trap in the Netherton Burn catchment. The widened ditch trapped sediment in an arable section of the catchment (the upstream area was 0.8km²). Figure 4.22 (left) shows the three-tiered sediment trap designed to slow the flow in the ditch, forcing suspended sediment and nutrients to be deposited in each of the storage 'cells'. Figure 4.22 (right)

shows evidence of flow attenuation resulting from the feature. In large storms, however, the overflow structures operated and the water flowed over the floodplain to the main ditch. This bypass flow did take a longer and rougher flow pathway but the amount of attenuation it provides was difficult to assess (Barber 2013).



Figure 4.22 Three-tiered sediment trap in Netherton Burn, Northumberland (left) and the impact of the ponds in attenuating the flow (right)

Source: Barber (2013)

Headwater peatland restoration

Peat covers 1.58 million hectares or about 7% of the land area in the UK, and is recognised as providing crucially important ecosystem services (Bonn et al. 2009). Although it is recognised that there may be opportunities for NFM measures in lowland raised mire and fen settings, the focus of this assessment is on upland peat management techniques. The process of peatland degradation is discussed in Section 4.1.5. Restoration measures have the potential to change the storage of water within the peatland and change the travel time of water through the system. This section focuses on 3 techniques:

- vegetation management
- moorland grip blocking
- gully blocking

Other techniques such as burning and grazing management are outside the scope of this literature review.

Vegetation management

There are a number of techniques for the restoration of vegetation on bare peat areas. Revegetating bare peat can significantly increase the roughness of the surface and thus reduce overland flow velocities (Holden et al. 2008). It is occasionally known for a nurse crop of grass to be used first to stabilise the peat surface before these techniques, such as Sphagnum spreading, are undertaken. The Yorkshire Peat Partnership provides guidance²² on a number of techniques:

- brashing spreading a 1cm thick heather or cotton grass brash mulch to provide stable soils and a microclimate for moss to establish
- dwarf shrub seeding in certain situations heather seeds can be spread over brash
- Sphagnum applications Sphagnum fragments or Sphagnum fragments contained within a growth medium pellet (known as BeadaMoss®) can be spread by hand or by soft track
- lime application where soil pH is below 4, even peat vegetation may be difficult to establish, so granulated lime can be applied to bare peat to raise the pH sufficiently
- heather bales can be installed across small overland flow routes to slow the velocity of water across the surface

The installation of geotextiles is also a technique that can be used (Moors for the Future 2008) to help re-establish vegetation in high energy drainage environments such as gullies. This is because, on steeply sloping peat systems, newly planted vegetation may erode away unless the peat is first stabilised and a nurse crop established.

Grayson et al. (2010) studied the impact on storm hydrographs of vegetation changes in the Trout Beck catchment in the north of England from the 1950s to the 2000s. The proportion of vegetated peatlands declined between the 1950s and 1970s and then began to increase. Peak storm discharges were significantly higher in the period of lowest vegetation cover and the time to peak shorter. However, total discharge was not affected suggesting that the most important control was on slowing flow rather than changes in the overall water budget.

Gao et al, (2016) developed a spatially distributed TOPMODEL to estimate the impact of changes in the spatial distribution of vegetation on peatlands. They showed that:

- The vegetation cover in the riparian zone and hill toe is critical for slowing the delivery of overland flow to the river.
- For the same total land surface area, bare peat patch size did not affect discharge rates. The potential increased connectivity of larger peat patches did not seem to have an impact at a catchment scale, as they might at a hillslope scale.
- Vegetation roughness has a larger control on run-off rates in flat areas rather than steep slopes. This may be because the travel time across gentle gradient slopes (all other things being equal) is longer than on steeper slopes, and so roughness effects are more important in gentle gradient areas.

Overall, Gao et al (2016) suggested than vegetation restoration and roughness increases should be targeted on riparian and gently sloping areas.

Moorland grip blocking

Moorland gripping consists of digging shallow ditches to drain wet areas of heath and blanket bog, and was particularly widespread in the UK uplands in the 1940s to the mid-1980s. The impact of drainage on the natural functioning of a peatland system is discussed in Section 4.1.5. Grip blocking is a process that has responded to gripping to restore natural drainage patterns, encourage revegetation, reduce erosion and minimise the knock-on effect

²² <u>http://www.yppartnership.org.uk/restoration/technical-guidance-notes/</u>

of hydrological change downstream. By increasing the vegetation in the uplands by grip blocking, the travel time can increase, reducing the time to peak flows and potentially increasing the number of flow paths to surface water bodies during storm events. However, in some circumstances, grip blocking can cause water to travel a shorter route across hillslopes to streams after restoration rather than longer contour-parallel grip channels, in which case the effects may be cancelled out.

A number of blocking techniques have been adopted. The Yorkshire Peat Partnership suggests a number of potential techniques for smaller grips:

- peat dams constructed from peat, these form by peat plug which rises above the top of grip and is stabilised by vegetated turfs (Yorkshire Peat Partnership 2014a)
- heather bales used as an alternative to peat dams (Yorkshire Peat Partnership 2014b)
- stone dams used in larger grips (between 1m and 2m wide) and in grips which have eroded down to the underlying mineral substrate (Yorkshire Peat Partnership 2014c)

Where grips are larger than 2m in depth or width, the Yorkshire Peat Partnership suggests that they are too large to dam. Reprofiling the sides to form a more stable profile is therefore recommended in these situations.

A result of these blocking techniques is the pools of water created behind each of these dams or bales contributes to additional flood storage space and acts similarly to dry retention ponds discussed in Section 4.4.

Gully blocking

Gullies are naturally occurring features of peatlands where blanket peats spread to the heads of valleys, although they also form where artificial drainage features become eroded. Other pressures such as wildfire, overgrazing or pollution exacerbate erosion by reducing vegetation cover. A report by the Joint Nature Conservation Committee (JNCC) report on the state of UK peatlands describes gullies as:

'fluvial erosion channels which cut into a peat mass, resulting in loss of peat and significant dehydration of adjacent in situ peat' (JNCC 2011).

As they are erosional features, they are often aligned –unlike grips – with the slope. Techniques for restoring gullies are often similar to large grips. Profiling, timber sediment trap dams and stone dams (among other dam-forming materials) are used within vegetation stabilisation techniques to stabilise gullies (Yorkshire Peat Partnership, undated). By blocking the gullies and encouraging vegetative cover within them, the travel time may increase and cause other flow paths to develop during rainfall events. However, the same issue applies as grip blocking in which the water could potential take a shorter route, cancelling the effects on flood flows. This technique, like grip blocking, also results in pools of water behind the features, which can contribute to additional temporary flood storage space (if the pools are able to drain down between each event).

Pilkington et al. (2015) sought to assess the impact of gully restoration measures, among a suite of other measures. They reported the following.

- Gully blocking did not have a statistically significant effect on reducing flood flows.
- The observed changes were consistent with the hypothesis that revegetation and gully blocking increased surface roughness. Surface revegetation reduces

overland flow velocities. Gully blocks and associated gully floor revegetation may also reduce in-channel velocities.

- Peat restoration by revegetation accompanying gully blocking has benefits for downstream flood risk reduction by 'slowing the flow' in peatland headwater catchments. Modelling is required to evaluate the benefits at larger catchment scale. This study provided robust empirical data and process analysis to calibrate such models.
- There was no change in percentage run-off within-storm events (that is, the proportion of storm rainfall producing discharge).

Modelling was carried out as part of this project and looked at implications of upscaling this work. Pilkington et al. (2015) concluded that gully blocking and vegetation restoration of 12% of the catchment would be associated with an average reduction in peak discharge of 5% for a 9km² catchment area. Overall, gully blocking did not have a significant effect. However, it was shown that, if different gully block designs (allowing some water to drain through them both during and after flood events) were used, then it could be possible to develop an attenuation effect on flood flows.

Lowland water level management

Lowland landscapes are complex areas that have different hydrological regimes and consequently different management requirements to other parts of catchments.

The need for drainage for productive lowland farming is high. Areas dominated by low topographic gradients often lack steep hydraulic gradients and so can be prone to waterlogging, particularly if soil permeability is low or if there are impeding/impermeable layers below the main soil layers. Thus many lowland agricultural soils have dense networks of underdrains and ditches to help drain water from the system to permit productive crop and grassland production. In some cases, ditch networks are also operated by pump systems as the topography is so gentle that ditches may not evacuate water from the landscape at a fast enough rate. Pumping from ditches is often required to draw down water levels, which in turn enables underdrains in the soil to function.

The loss of organic soil material and soil shrinkage following the lowering of the water table has often caused the surface land levels to fall, lowering drainage gradients even further and particularly in lowland peat soils used for agriculture (Lindsay et al. 2014b). However, these soils are some of the most productive in the UK when carefully managed (ADA 2015). Lowered water table and ditch water levels enhance temporary flood storage in these systems. In some systems, however, the flood threat to agriculture means that measures involving moving the water rapidly out of these areas are often undertaken which may exacerbate flood risk downstream. The requirements for localised water and drainage management in these complex lowland locations led to the formation of Internal Drainage Boards (IDBs).

Many of the fundamental functions of IDBs have been an integral part of water level management in the UK since the 13th century. Each IDB is a local public authority established in areas of special drainage need in England and Wales. IDBs have permissive powers to manage water levels within their respective drainage districts. They undertake works to reduce flood risk to people and property and manage water levels to meet local needs. IDBs are concentrated in the Broads and Fens of East Anglia, Yorkshire, Somerset Levels, Kent and Nottinghamshire (ADA 2015).

Lowland ditch networks, especially in productive agricultural landscapes, are highly managed and controlled environments, with maintenance including vegetation cutting/removal and dredging (see Figure 4.23). Detailed advice, guidance and information

about funding resources relating to lowland ditch management regimes are available from the website of the Association of Drainage Authorities (ADA).²³ Water level control through the operation of the ditch network is the principal land management objective. It includes:

- movement of water
- supply of water to groundwater and irrigation
- control of environmental conditions such as for fenland and washland for habitat purposes



Figure 4.23 Lowland ditch management options: vegetation cutting (left) and sediment removal (right)

Source: ADA

Both ADA and the National Farmers' Union (NFU) have agreed to support lowland ditch network management to help deliver flood management objectives, as seen in the recent NFU floods manifesto (NFU 2017).

Drainage systems in IDB areas can be managed to improve flood resilience. In addition to vegetation and silt management, water levels for drainage and irrigation needs can be varied artificially over the year by the operation of control structures and pumping stations. Water levels can thus be set at lower levels during periods of increased flood risk to increase the capacity of the drains (ADA 2015). Flood forecasting models, based on predicted rainfall, can allow drainage systems to be pre-emptively pumped in advance of a rainfall event (ADA 2015).

Brompton in Yorkshire lies within the Swale and Ure IDB and was subject to severe flooding in 2000 and 2012 (Metcalfe et al. 2017). This is an area of very high agricultural productivity, and has been drained intensively by the local IDB for many years to maximise the conveyance of flow through the drainage network. This may have inadvertently helped to increase the flood risk to vulnerable downstream communities. A partnership approach is now underway with the IDB and Yorkshire Dales Rivers Trust to manage the run-off rates from the ditch network without compromising the local agricultural activities. In Figure 4.24 shows the construction of leaky dams in the ditch network. The theory is that large amounts of water can be slowed and stored in the ditch network itself. Metcalfe et al. (2017) used a modification of TOPMODEL to simulate both the ditch network and potential impact of the installation of a network of within-ditch leaky barriers (see Brompton Catchment Case Study). The features were designed to let the normal (non-flood) flows under them, as they were usually set 30cm above the bed of the channel. Sufficient space was created above the feature to allow overflow to occur and flow was not allowed to enter the surrounding fields

²³ www.ada.org.uk

due to the perceived negative impact to local agricultural activities. The model simulations allowed for up to 60 within-ditch features. The findings suggest that:

- it is vital to understand the timing and synchronisation of the flow from the established ditch network
- the smaller features upstream were quickly overwhelmed and therefore had less effect on the peak flow

In the Brompton area, new locations for the storage of flood flow in offline ponds are now being sought as an addition to the leaky dams.





Source: Metcalfe et al. (2015)

Typical water management options that are found on lowland ditch systems are either water retention or water removal. In the Great Fen restoration project in East Anglia,²⁴ a number of fields are moving away from intensive arable production to wetland types, primarily for environmental improvement but delivering multiple benefits including flood attenuation. Raising water level in ditches (dykes) can also create wet meadows, which has a high biodiversity value. Figure 4.25 (left) shows an example of a sluice in a ditch that can back up water into fields. It is becoming common practice in ditches for the banks to be regraded and new vegetation habitats encouraged (Figure 4.25 right).



Figure 4.25 The Great Fen showing (left) a typical sluice for maintaining the water level in a ditch and (right) an example of steep sided ditch that has been regraded

Source: Great Fen Team (undated)

Together the lowering of farming intensity and the slower movement in ditches should lower discharge rates. However, there is no solid evidence as to the likely impact on flood

²⁴ http://www.greatfen.org.uk

reduction. The Environment Agency and the Middle Level Commissioners are working to create new larger floodwater storage areas in the Great Fen area for times of heavy rainfall.

In the Somerset Levels, recent flooding (2013 to 2014) sparked a debate about management of the lowland ditch network. Many farmers work closely with the IDB and are paid through agri-environment schemes to keep the water levels high during specific periods of the year so as to deliver environmental benefits. In response to the recent episodes of prolonged flooding in the Levels, dredging was commissioned at many locations to increase the conveyance rate of the ditches and reduce the duration of flooding. However, some commentators have suggested that this means that the modified ditch network could make flood impacts worse downstream (CIWEM 2014).

A report by the Chartered Institution of Water and Environmental Management (CIWEM) concluded that dredging is not a single solution to flooding in these lowland areas (CIWEM 2014). The report noted that studies over the past 30 years had indicated that dredging can speed up flow, with potentially increasing the risk of flooding downstream and having detrimental unintended consequences for wildlife, properties and businesses. However, the report recognised that dredging does have a role in flood risk management in certain circumstances but, by itself, cannot prevent flooding during extreme rainfall events (CIWEM 2014). The need for dredging and channel maintenance (Environment Agency, 2015) may be justified in IDBs which deliver regionally/national important food production services. However, the need to take a catchment-based approach has been recognised by the 20-year flood action plan (Environment Agency 2015d).

The complexity of flood mechanism in this landscape setting means that the topic of applying NFM measures in this environment has not been well explored. The main focus of this section is therefore on upper catchments and hillslopes, and lowland measures are not discussed further in this literature review.

4.3.3 Confidence in the science

The design features discussed in this section on land drainage and headwater management can all help slow the flow, or change the storage of water within the landscape. However, there is little quantitative evidence that they affect flood risk at the larger catchment scale. It is likely that there is an aggregated effect downstream, but it is not readily detectable using current datasets.

Monitoring of measures is most easily implemented on a local scale, whereas modelling can be done on a range of scales from individual measures to catchment modelling of the aggregated impacts of a spatial array of measures. The confidence in scientific evidence consequently varies between scales.

There is a plethora of modelling studies that attempt to simulate change (Environment Agency WWNP, 2016a) and the FEH presents some standard methods for catchment simulation. The modelled evidence to determine the impact of land and headwater drainage management can be determined by the following concepts which underpin the basic hydraulic formulae:

- cultivation features
- grip blocking
- ditch management
- hydraulic roughness (Manning's n)
- size and capacity of the ditch

• obstacles to back up flow and induce sedimentation

Environment Agency (2016a) provided information on the suitability of different modelling approaches and the decision-making process behind selecting models to address a range of scenarios. For catchment modelling, McIntyre and Thorne (2013) provided a detailed discussion on the range of modelling approaches that can be used, from lumped models which represent a catchment with one average set of model parameters, to fully distributed models which take into account the spatial patterns of hydrological response within catchments. A synthesis of the types, input requirements and outputs of different catchment modelling approaches is shown in Figure 4.26.

Model type	Lumped				Semi-distributed		Fully distributed
Model name	Rules of thumb	FEH	CFMP	FRMD	Regionalised runoff	Physics-based runoff	Fully distributed physics-based models
					Continuou	is rainfall and ET	
Input requirements		FEH CD		н	OST	Physica	I properties
		Land cover (%)		Spatially distributed land cover			
						Digital elevation model	
					Base	eline streamflow observa	tions
Outputs	Direction				Change in flood frequer	ncy curve	
	of change			Complete hydrograph			
Modelling expertise	LEAST						MOST
Computer power	LEAST						MOST
Potential Reilability	LEAST						MOST

Figure 4.26 Types, inputs and outputs of catchment modelling approaches

Source: McIntyre and Thorne (2013)

There are numerous models available to simulate hydrology and the hydraulics at a landscape/catchment scale such as Flood Modeller, HEC-RAS, MIKE FLOOD and TUFLOW. Models often break down into 1D and 2D model types depending on how accurate the study, together with the quality and quantity of model input data. Estimates of parameters can be made and design storms can be simulated. However, at the catchment scale, the empirical data (of the landscape parameters) to be inputted into the models have extensive variability, as does the spatial variability of the rainfall, which confound the identification of evidence of change that can be directly attributed to the WWNP measures. Approximation of run-off and roughness that are not calibrated for the specific catchment study areas and the accumulation of errors in models can lead to a poor model uncertainty estimates (Beven 2010, McIntyre and Marshall 2010). Even in longer term time series, it is difficult to detect and quantify change in areas that have undergone land use change and management (Beven et al. 2008). A fuller discussion of modelling of intervention measures is presented at the end of this section (see Section 4.3.5).

Agricultural headwater drainage

There is good evidence that altering the hydraulics of a ditch will lower flow rates at the intervention site. However, these features have often not been installed primarily for flood flow retention and hence understanding of the operation of these features in flood events remains weak as shown by monitoring and modelling results. Even those ditch modification features that have been installed primarily for flood flow at Nafferton Farm have not been monitored for long enough to show performance in a real flood flow event. However, some confidence can be built locally based on the observations of a feature. Observation and the interplay with model construction simulations can extend the range of operation with some

confidence. There is some confidence that the nature of the intervention is positive to flood risk management and hence the confidence is classed as medium level at the local scale.

There is limited confidence on a local scale because the evidence of interventions relies on studies primarily focused on diffuse pollution management and there is very little quantifiable evidence from that those specific to flood attenuation. There is some scientific evidence from studies such as those by Whitworth (2011) and The Rivers Trust (2014), which concluded that increasing hydraulic roughness – through vegetation – in ditches can reduce flood flow. However, a large amount of evidence is qualitative with limited semi-quantifiable evidence found in case studies such as the Eden DTC study, the study at Nafferton Farm and the Environment Agency funded project delivered by the Arun and Rother Rivers Trust (Wright 2014) which support a storage and attenuation effect to varying degrees. The results from the Arun and Rother Rivers Trust study demonstrate how the confidence of widening the ditch can vary depending on its design. This is further discussed in Sections 4.3.4 and 4.3.5.

Additionally, in the Nafferton Farm study, a number of in-ditch features were simulated. A 1D hydraulic model (Kutija and Murray 2007) was used to estimate the impact of widening and flattening a small ditch and roughening the vegetation. As can be seen in Figure 4.27 widening and roughening the ditch contributed to a reduction in flood peak, helping to slow and store flood water.

In the Nafferton Farm project (as reported in Jonczyk et al. 2008), a number of within-ditch features were built and tested. The features targeted sediment and phosphorus reduction. To maintain the function of the feature during larger storm events, the barrier was designed to attenuate large amounts of flow. However, it was difficult to estimate the flow through the barrier and the residence time; it was estimated to be 30 minutes according to Palmer (2012). Ockenden et al. (2012) presented the results from a number of ditch features, which they referred to as 'in-line wetlands' in modified ditches. All 10 of the constructed features showed a good reduction in phosphorus and sediment levels, and the associated potential flood attenuation benefit was inferred rather than measured.



Simulation duration of 1 hour

Figure 4.27 1D hydraulic model of a widened and roughened ditch, Nafferton Farm study

Source: Quinn et al. (2008c)

Headwater peatland restoration

Vegetation management

Surface cover in peatlands is a significant factor influencing overland flow velocities (Holden et al. 2008). Grayson et al. (2010), Pilkington et al. (2015) and Gao et al. (2016) provide data based on monitoring observations and modelling approaches suggesting that vegetation cover and management can increase the time to peak and reduce peak flow. Sphagnum vegetation provides significantly greater roughness to overland flow than *Eriophorum* (cotton

grass) or bare peat (Holden et al. 2008) and therefore increases the travel time of water within a peatland. The spatial distribution of measures controls the effectiveness, with revegetation measures within the riparian zone being most effective (Gao et al. 2016).

Grip blocking

Based on the evidence, the confidence in grip blocking on a local scale in reducing flood risk is variable.

The impact of grip blocking has been shown in numerous studies (Holden et al. 2008, Wilson et al. 2010, Holden et al. 2016); the results are generally positive with both a short-term and long-term recovery of water tables, the creation of desirable vegetation (namely Sphagnum moss) and the slow infilling of the local pond created behind each dam. However, it is less clear that there is a significant impact on flood flow or on the downstream water resources. Depending on the situation, grip blocking can increase or decrease discharge rates at a hillslope scale. The controls on this and the uncertainty in the response are discussed in Section 4.3.2. Consequently, the effects of grip blocking on run-off is site-dependent and relates to the orientation, density, and topographic context (Lane and Milledge 2013).

The pools of water created behind each dam may not have a significant amount of storage attenuation, especially during storm events, since grip blocking often raises the water table. The redirection of water and the changes in travel path length and travel time are therefore more crucial in impacting flood flows, but this is still obviously dependent on the site.

Wilson et al. (2010) studied the effects of grip blocking in a small catchment (12.5ha) and found that blocking quite rapidly resulted in more seasonably stable and marginally higher water tables, enough to increase the generation of saturation overland flow. However, the rate of response of the catchment to rainfall was reduced, with decreases in the 1 percentile exceedance flow observed, based on one year of data pre blocking and one year of data post blocking. The analysis suggested that the reduction in rate of response was a net effect of a reduced drainage density (Wilson et al. 2010).

Holden et al. (2016) also monitored ditch blocking. They identified that in the first year, there was a 5-fold reduction in discharge down the ditch blocked drains, but in the 5 subsequent years, the discharge rates doubled from the initial low point, indicating that the effectiveness of restoration measures is not static as soil properties may change over time in response to restoration.

The conclusion from the Lane and Milledge (2013) modelling study was that surface roughness on hillslopes is critical and important to determining the impact the speed of hydrological responses on upland areas.

The study by Ballard et al. (2012) used a simplified physics-based model to produce evidence about and uncertainty analysis of the effects of drainage management on flood peaks. The model showed that the drainage of peatlands will increase peak flows and that drain blocking will usually decrease peak flows but may actually increase them in some cases. The analysis suggested that greater reductions in peak flows following drain blocking may be observed with time as surface roughness increases from the recolonisation by rougher peatland species. However, the magnitude of these changes will depend on the degree of temporal analysis, the degree of recolonisation, and the state of the vegetation prior to drain blocking (Ballard et al. 2012).

In contrast to the monitoring study by Holden et al. (2016), current modelling studies do not appear to consider the potential long-term impact of grip blocking, that is, vegetation changes to assemblages more akin to intact peatland and revegetation and eventual infilling of the remnant grips (aided by reprofiling). Grip blocking as part of a suite of restoration measures may, in certain areas, lead to an approximation of the hydrological regime of an

intact peatland. The long-term impacts of grip blocking and restoration works in general may be very different to the initial short-term effect.

Lane and Milledge (2013) developed and applied a model for assessing the impacts of grips on flow hydrographs. Their simulations suggested that the delivery effects may not translate into catchment-scale impacts for the following 3 reasons.

- The proportion of flow path lengths that were hillslope were not changed significantly by gripping.
- The structure of the grip network compared with the structure of the drainage basin mitigated against grip-related increases in the concentration of run-off in the drainage network.
- The effect of a marginal reduction in the mean timing of run-off at the catchment outlet can only be assessed by reference to the peak timing of other tributary basins, emphasising the drain effects are both relative and scale dependent.

Gully blocking

There have been limited studies into the impact of gully blocking on run-off rates to determine the confidence of gully blocking in reducing flood risk. Pilkington et al. (2015) is an important study that showed that, from empirical data, the storm run-off peak was not significantly affected. However, the modelling suggested that there could be a long-term flood attenuation effect once the measures are fully bedded in and mature, and that it would be a stronger effect if the gully blocks were redesigned. Lastly, Pilkington et al. (2015) determined that gully blocking did not produce a significant decrease in run-off rates compared with gullies; however, gully blocking was not statistically significant when compared with the impacts of the revegetation of bare peat alone on reducing the run-off.

Modelling approaches

This section provides an introductory discussion on empirical and modelling approaches, in relation to ditch interventions, on a local and catchment scale.

Manning's n and hydraulic parameters such as slope and wetted perimeters can all be used to calculate discharge rates in ditches, and approximations relating to ditch management can be made locally. There is confidence that a well-designed ditch with a well-designed mitigation measure will operate as designed. However, at the catchment scale, the effects are more difficult to determine and hence the need to rely on model outputs.

Standard flood estimation methods can be used to quantify the likely run-off from a catchment for a given return interval design storm. There is good software support for these tools and the parameters listed can be accepted on a geographical basis. The good experience of practitioners of following these approaches and similar models is detailed in Environment Agency (2016a).

Other designs are based on standard engineering methods and empirical equations. These equations can be used directly to estimate the capacity required for a determined flow rate, and hence the size and form of a ditch can be designed. The impact of ditch form and roughness can be tested using these equations. It is also possible to build in a grip or ditch drainage network to a distributed model.

The new spatially distributed version of TOPMODEL has an overland roughness module to simulate the impact of roughness on overland flow pathways, while SCIMAP²⁵ (underpinned by TOPMODEL routines) examined the impact of the grips on run-off production. For example, Appendix F of a report by the River Ribble Trust reported on SCIMAP modelling

²⁵ <u>http://www.scimap.org.uk</u>

used to compare a series of intact and gripped catchments in the Upper River Ribble Study (River Ribble Trust 2015). Gao et al. (2016) applied a spatially distributed version of TOPMODEL to the Trout Beck in Cumbria, the Wye basin in mid Wales and the East Dart basin in south-west England. They showed that the model performed well and that scenarios comparing bare peat with restored peat (rich in Sphagnum) showed significant reductions in flow peaks.

Finally, the effects of ditch blocking in upland peat and agricultural systems on groundwater storage and surface water routing can be quantified by modelling. Studies suggest that it is generally successful in raising the water table (<0.1m) in the immediate vicinity of the drains (LaRose et al. 1997, Worrall et al. 2007a, Wilson et al. 2010), although it may not rise to the levels observed for intact peat sites (Holden et al. 2011). Ballard et al. (2012) showed through modelling that drain blocking does not recreate the hydrological response of intact peatlands; blocked drains consistently produced higher peak flows than intact peatland. This is explained by the fact that blocked drains often overtop in locations that concentrate flow depth over the land, reducing the effect of hydraulic roughness (Ballard et al. 2012), as seen in Figure 4.28.



Figure 4.28 Schematic representation of run-off generation in a (recently) blocked grip prior to, early on in or after a rainfall event (left) and during a rainfall event (right)

Source: Geris (2012)

4.3.4 Metrics of cost–benefit and multiple benefits

Land and headwater management studies often focus on other benefits rather than reducing flow and flood risk. Consequently, there are known multiple benefits to installing different measures of land and headwater management. Other benefits include reducing water pollution, improving ecological diversity, and reducing erosion and sedimentation. Peatland restoration also generates a carbon sequestration benefit.

Agricultural headwater drainage

The most studied benefits include reducing sedimentation and pollution from surface water run-off. In the Nafferton Farm study (as reported in Jonczyk et al. 2008), a number of withinditch features were built and tested with the aim of reducing sediment and phosphorus levels. Ten features were constructed in this Defra-funded study, which all showed a good reduction in phosphorus and sediment levels (Ockenden et al. 2012).

The widened ditches to form online sediment traps were monitored as part of the Cheviot Futures project in the Netherton Burn catchment (Barber 2013). The widened ditch trapped sediment in an arable section of the catchment (the upstream area was 0.8km²). During storm events, there was qualitative evidence that the overland flow, or bypass flow from the overflow structures, took a longer and rougher flow pathway and had a strong impact on water quality. The evidence looked at paired samples collected from the inlet and outlet determinand concentrations (Table 4.3). It concluded a significant mean concentration reduction percentage for all determinands (Barber 2013).

Determinand	n	Mean % reduction	Paired T-Test	
			T-value	P-value
SS	174	42.4	11.53	<0.001
ТР	174	25.7	13.88	<0.001
SRP	122	14.9	7.22	< 0.001
NO ₃	108	5.2	7.01	<0.001

Table 4.3Reduction in pollution in the three-tiered pond monitored as part of the
Cheviot Futures project

Notes: Mean percentage reduction and significance for 174 samples across 9 storm events. SS = suspended sediment, TP = total phosphorus, SRP = soluble reactive phosphorus, NO₃ = nitrate Source: Barber (2013)

The research by the Arun and Rother Rivers Trust on widening ditches as sediment traps looked into the effectiveness and practicalities of sediment traps as a means of reducing sediment-laden run-off into the western Rother catchment (Wright 2014). The main conclusion from this research was that it was both uneconomic and impractical to construct sediment traps that attempt to catch the majority of surface water run-off within a watershed area since the traps would have to be unfeasibly large. However, it was concluded that smaller traps are still effective in trapping the larger fraction of suspended sediments in surface water run-off from between 63µm and 500µm in diameter.

Traditional metrics such as peak discharge or time to peak change have not generally been used in the studies reviewed here as they were either not the focus of the study or the evidence could not be upscaled to the catchment scale. Apart from bold changes to FEH parameters, there is currently no catchment-based approach for headwater drainage where the allocation of model parameters has very high confidence. Therefore, metrics such as the level of protection and properties saved are subject to great uncertainty and many assumptions.

There are many other benefits to good ditch management (AHDB 2015, Shore et al. 2015). However, when performing a cost–benefit analysis of these drainage features to reduce flood risk, including capacity and conveyance characteristics, a number of additional factors need to be balanced against each other. The long-term maintenance of agricultural productivity in the fields draining to these ditches is a major factor that needs to be considered.

The cost of these interventions is fairly low and economical. The cost of ditches and ditch maintenance aligns primarily with agronomic reasons. So logically ditch interventions and management will have similar costs for flood mitigation (AHDB 2015, ADA 2015). It is always difficult to estimate costs as each field is different and the balance between underdrainage, subsoiling and the ditch construction itself varies. In the example given in AHDB (2015), a typical 10ha field cost £545 per hectare for underdrainage and £5,000 of ditching work (so ~£500 per hectare). The payback time to the farmer was expressed in improved yields, which took 13 years to pay back. The additional annual management costs might be low, but for some, the long-term effectiveness depends on maintenance and eventual replacement when the asset is worn out/damaged. It would be hoped that aquatic vegetation management would equally fall into this category. In Environment Agency (2017), numerous examples of cutting and species management all added to the annual cost. However, many farmers are working with non-governmental organisations (NGOs) and the Environment

Agency to manage in-channel vegetation more sustainably. Increasing roughness through vegetation is also an economical intervention to slow the flow. In the Nafferton Farm study, adding a variety of willow cost £500 for a 30m section (Quinn 2007a).

Similar cost and benefits terms are needed to add a 'flood measures on farms' term when appraising these measures. This is subject is still in its infancy, though methods using ecosystems services have some potential (POST 2015). Even though the numbers used in the calculation may be subjective and subject to uncertainty, it is still possible to estimate cost and to rank relative cost of activities. Equally, the value of environmental activities can also be approximated (Ozdemiroglu and Hails 2016), with efforts being made to determine how much worth something has to people or society. Concepts such as well-being, welfare and utility are often used as value terms. These terms must be counteracted by a willingness to pay (Ozdemiroglu and Hails 2016). Hence if a better argument can be made to the flood risk benefits of NFM measures, then a broader economic assessment of funding could be made that society may be willing to pay for.

Headwater peatland restoration

Capital costs of peatland restoration vary with degradation state and the ease of access to the site. Moxey and Moran (2014) attempted to quantity the economics of peatland restoration. They assessed that capital costs fall between £200 per hectare and £10,000 per hectare. In addition, there are costs associated with monitoring and maintenance. Opportunity costs also have to be considered, as restored peatlands can often replace profitable activities such as forestry. Together these total additional costs were assumed to range between £25 per hectare and £400 per hectare.

Multiple potential benefits accrue from peatland restoration (Moxey and Moran 2014) including:

- water quality management in treating water discolouration
- habitat and biodiversity improvements (Ramchunder et al. 2012)
- protection of archaeological and paleoenvironmental features
- recreational and landscape benefits

Peatland restoration also can bring carbon benefits. Degraded peatland release the carbon stores created during periods where the peatland was actively growing. However, calculating the carbon benefit of the peatland restoration is complex. For example, rewetted peat can increase the emissions of methane, which can partly offset the lower carbon emissions. Moxey and Moran (2014) estimated that peatland restoration could bring differential benefits of between 1 and 20 tonnes of carbon dioxide per hectare per year, and depending on the value of carbon, could in many cases be more valuable than the capital costs. However, this conclusion was based only on the value of carbon savings and did not quantify the other potential multiple benefits.

4.3.5 Effectiveness/performance

Agricultural headwater drainage

There is good evidence that altering the hydraulics of a ditch will lower flow rates at the intervention site. However, the interventions that have been installed primarily for flood flow do not provide evidence of the effectiveness or performance on flood flow.

The trialling of widening the ditch to create in-ditch sediment traps by the Arun and Rother Rivers Trust evaluated the effectiveness over a 3-year period by monitoring the amount of sediment accrued by the traps over time and preventing the surface run-off from entering local rivers (Wright 2016). The study concluded that constructed sediment traps that attempt to catch a majority of surface water run-off within a watershed is impractically and unfeasibly large (Wright 2016). Smaller sediment traps, however, are still effective in trapping a larger fraction of suspended sediments in surface water run-off.

The monitored evidence for the effectiveness and performance of widening the ditch in a flood risk management context is extremely limited. There is, however, stronger modelled evidence. The Nafferton Farm study (Quinn et al. 2007a) showed that vegetating ditches reduced peak discharge (Qp) by 3.86%.

Levasseur et al (2012) addressed the density of ditches on run-off response in agricultural systems. In the model, the number and length of the ditches were captured explicitly and the ability to capture overland flow pathways was simulated. The results highlighted the importance of spatial configurations of agricultural ditch drainage networks in the alteration of water flow paths and in the control of run-off, both in channelised flow paths and on hillslopes. In addition, the case study presented by Levasseur et al. (2012) showed that the delineation and area of the subcatchment were greatly influenced by the spatial configuration and increase in the ditch network density, which increased the drained volume and the peak discharge, decreasing overland flow on hillslopes. However, these hydrological behaviours were only sensitive to network density for drainage networks with densities that did not exceed a certain threshold (Levasseur et al. 2012).

At a catchment scale, modelling studies such as the one by Metcalfe et al. (2015) indicate that small features can easily be overwhelmed, with their flood risk management benefits decreasing downstream. Issues of synchronicity mean that their placement in the system can be important in controlling their overall effect.

Headwater peatland restoration

Vegetation management

There is significant evidence at a range of scales that restoration techniques which replace bare peat with vegetation can reduce run-off rates through increased hydraulic roughness.

The natural catchment experiment by Grayson et al. (2010), which compared hydrograph response since the 1950s to the changes in the bare peat coverage over the same period, showed that when bare peat coverage was at its maximum the flow hydrographs were significantly peakier with higher peaks per unit of rainfall (0.40m³ mm⁻¹ compared with 0.27m³ mm⁻¹ for a period of good vegetation).

Holden et al. (2008) presented field data on the velocity of overland flow and drain flow in upland peatlands, exploring the relationship between flow velocity, vegetation cover, slope and water depth. The mean overland flow velocity was significantly higher for bare surfaces than for vegetated surfaces for all discharge categories. The velocities associated with Sphagnum were significantly lower than other vegetation types, suggesting that Sphagnum is better at attenuating flow velocities than the other cover types tested. The dense branching and uniform structure with depth can have implications for overland flow travel times, potentially leading to shortened stream lag times (Holden et al. 2008).

Pilkington et al. (2015) conducted an intensive field monitoring campaign from 2010 to 2014 with a pre- and post-restoration study of degraded micro-catchments on Kinder Scout in the Peak District, with additional data from established reference sites. The results showed statistically significant peat restoration-related changes as the catchments became wetter following revegetation, as evidenced by the water tables rising by 35mm and overland flow

production increased by 18%. The storm flow lag times in restored catchments increased by up to 267%, while peak storm discharge decreased by up to 37% and the hydrograph shape index reduced by 38%. However, there were no statistically significant changes in percentage run-off, indicating limited changes to within-storm catchment storage to date. The results demonstrated that storm water moves through restored catchments more slowly, attenuating flow and storm hydrograph responses to potentially benefit downstream flood reduction. However, it is not clear from the empirical evidence outlined what impact this revegetation would have on a catchment-scale discharge and downstream flood risk (Pilkington et al. 2015).

Modelling by Gao et al. (2016) identified that replacing bare peat along a riparian zone with Sphagnum could reduce peak flows by between 1.8% and 13.4% flows for a 20mm per hour event.

Grip blocking

The evidence for the effectiveness of grip blocking on storm peaks is inconsistent (Shepherd et al. 2013), as grip configuration and topographic context are critical.

Consequently, the effect of grips and grip blocking on run-off is site dependant. Lane and Milledge (2013) identified that drainage should provide drier antecedent soil moisture conditions and thus provide more soil water storage. Although the orientation of grips relative to the hillslope can also mean that they intercept run-off flow paths, the velocity of flows within grips can be higher than overland flow. As a result, grips have the potential to increase or decrease peak run-off rates at a local and catchment scale.

If grips have the potential to increase or decrease peak run-off, this means that the impact of grip blocking is partly dependent on the effect of the grips being blocked. Ballard et al. (2012) identified by modelling that the blocking of steep smooth drains would most likely show the greatest reduction in peak flows following drain blocking. Furthermore, a series of model simulations at a field scale identified that drainage was generally found to increase peak flow, but that the effect of drain blocking was affected by local conditions and could lead to an increase or decrease in peak flow. They identified that the blocking of steep smooth drains are most likely to have the greatest impact on reducing field scale peak flow (Ballard et al. 2012). As previously stated, greater reductions in peak flows following drain blocking may be observed with time as recolonisation of rough peatland species occurs, although the magnitude of these changes will depend on the degree of recolonisation and the state of vegetation prior to blocking.

Additionally, grip blocking does not initially lead to conditions equivalent to intact peat. Water can still flow down grip channels, reach blocks and spill onto the surrounding land. This leads to a concentration of overland flow at these spill points (Ballard et al. 2012).

Gully blocking

There have been limited studies to assess the impact of gully blocking on flow rates. Pilkington et al. (2015) found no significant effect of gully blocking on peak flows, but modelling suggested that there should be a flood attenuation effect and models of different block designs suggest that NFM benefits could be enhanced.

Overall

With the exception of vegetation management, there is not much direct evidence for the effectiveness of peatland restoration measures in reducing either run-off rates or flood risk management. Peatland restoration techniques are often implemented as a suite of measures, and in many situations, the main aim of gully and grip blocking is peat stabilisation. If this is the case, they can be viewed as measures to limit degradation and

erosion, and the creation of areas of bare peat. As a result, they could be as important in the long term at direct vegetation management techniques in managing run-off rates.

4.3.6 Key case studies

This section selects a series of key case studies from those discussed above.

Moors for the Future and the associated review of peatland research (see, for example, Pilkington et al. 2015) has given the flood risk management community confidence in both process understanding and in practical methods to restore peatlands. The potential to slow and store flow is highlighted. This is a valued landscape, and the cost and effort required to implement measures appear to be justifiable. More work will be needed to prove that slowing flow in the peatland is having tangible flood benefits downstream. However, the process-based knowledge gained seems to suggest that lower connectivity and forcing longer, slower flow pathways should be beneficial.

The **Eden DTC** in Cumbria (Eden DTC 2017) highlights the work of similar projects in addressing farming and pollution. In the Morland mitigation subcatchment, some 20 management actions have occurred across 2 farms, tracks, roads and ditches in an area of up to 2km². Although it is difficult to quantify the actual impacts, a number of pollution hotspots – linked to surface run-off – have been disconnected. Soil and slurry management has improved and soil aeration is underway. Several known polluting pathways have been diverted onto grass fields and into sediment traps. The flow has been slowed with the construction of numerous ditch barriers and larger ponds. Uptake by the farmers has been good as they understand what is being achieved. The only tangible metric so far is the successful trapping of sediment in the ponds and ditches.

Brompton in north Yorkshire is an example of a small community at risk of flooding that has an IDB area immediately upstream. Efforts have been made to convey flood flows through the town more effectively, but an upstream WWNP solution is now being trialled. A number of leaky wood barriers are being placed into the ditch network as a demonstrator to farmers. Though modelling, Metcalfe et al. (2015) showed that:

- certain locations for these measures may be better than others
- care must be taken to avoid potentially generating detrimental synchronisation issues

More recently, the Flood Action Group at Brompton is looking for locations to store more flood flow in temporary offline ponds in fields.

4.3.7 Design, management and maintenance

Agricultural headwater drainage

Basic designs for cross drains are not complicated,²⁶ but more complex designs are possible. These are well described in the CIRIA SuDS Manual (CIRIA 2015) or on the Susdrain website.²⁷ The shape and characteristics of the catchment are of particular relevance to drain blocking activity, as they control the degree and timing of run-off concentration through a network effect, and can influence the timing and concentration of flow from different areas (Lane and Milledge 2013). In addition, the 'Channel Management Handbook' (Environment Agency 2015e) proposed that ditches are improved in function if:

²⁶ See the Farming and Wildlife Advisory Group (FWAG) website (<u>http://www.fwag.org.uk</u>).

²⁷ http://www.susdrain.org

- the channel banksides are regraded to 45° degrees
- rough vegetation is encouraged and also maintained appropriately

However, a study by Shore et al. (2015) which investigated ditch size and function concluded that there is no real design criterion for the size of the ditch and often the ditch is oversized for the area of land it is meant to drain. However, the potential to use oversized ditches for slowing flow either by obstruction and or encouraging rough in-channel/bankside vegetation is high. A number of other in-channel options have been trialled to help slow peak flows (Figure 4.29.



Figure 4.29 A trial leaky dam in the ditch at Nafferton Farm (left) and Belford Burn using a series of wooden flow restrictors (right)

Source: Quinn et al (2007a), Wilkinson et al. (2010)

Even though these structures were built in an agricultural ditch, the local Environment Agency officers suggested that flow under the structure would be preferred. Hence, when similar features were implemented at Belford (Wilkinson et al. 2010, Wilkinson et al. 2015), the base of the barriers were placed 30cm above the ditch bed to allow baseflows to pass under the barriers unhindered. The structures would back up flow at higher flows and then overtop, but not flood the surrounding fields. The cost of the nest of barriers was £9,000 as the barriers were built to a very high safety specification. The Environment Agency would recommend the use of barriers with good dry weather flow bypass underneath. This does lower the sediment trapping capability but perhaps sediment traps may be better applied in the run-off source fields (see next section) or in ditches on farms where there are no fish bypass issues.

Headwater peatland restoration

Peatland restoration techniques have evolved over the past few decades. Organisations such as Moors for the Future have monitored the effectiveness of a range of techniques (see, for example, Evans et al. 2005), Buckler et al. 2013).

The Yorkshire Peat Partnership provides on its website²⁸ a series of technical guidance notes for implementing a range of techniques including:

- blocking with peat dams
- blocking with heather bales
- blocking with stone dams

²⁸ <u>http://www.yppartnership.org.uk/restoration/technical-guidance-notes/</u>

- blocking with timber and composite dams
- peat hag and gully reprofiling
- bare peat revegetation

Outline costs for capital and ongoing works for peatland restoration are discussed in Section 4.3.4.

4.3.8 Research gaps

Better performance indicators are needed that show that value of investment in NFM measures or in measures that have associated NFM benefits. Natural capital valuation systems or ecosystem costing may help increase confidence in the take-up of these measures.

Given the problem of quantification and cost at the catchment scale, a much greater emphasis is needed to show why features work and how they perform in large events.

More research is needed to determine design specifications and guidelines. For example, there is evidence on how the design of gully blocking and the structure of the gullies can determine the effectiveness and performance in reducing flood risk. Further research is needed to test which gully blocking design can maximise NFM benefits.

It may be possible to see why interventions work and optimise them to operate effectively during flood flows, especially when overland flow paths are known, if nearly all the flood flow will be passed through the ditch network before being passed on downstream to larger catchments.

Further research needs to consider the spatial and temporal elements, in particular for drain blocking or peatland restoration. For example, there may be slow or gradual shifts in the system in response to drain blocking and short-term monitoring programmes will not pick up these subtle changes. This highlights an important gap in restoration monitoring.

A large research gap identified from this literature review and seen across many interventions to improve run-off management is how to determine the impact on flood risk management on upscaling or how to detect the impact at catchment scale. To upscale impacts, research needs to expand their study area from headwater to downstream using nested data. These nested observation networks in place from hillslope to catchment scale can test the spatial models that have been developed and create more appropriate parameters. There also needs to be methods that work better at detecting impact at the catchment scale. However, this may not be easily achieved given the quantitative uncertainty of observing and modelling at this scale. A better risk-based model may be needed.

Furthermore, the number and combination of interventions need to be further explored to determine an optimal number and design to provide multiple benefits and some specific flood reduction capacity. The ability to simulate features and clusters of features remains a research gap. The ability to add these local impacts to catchment model needs to be clear and transparent, and the uncertainties considered and reported. If this modelling is done better, then the risk of any investment made upstream may be matched by confidence that flood risk management benefits are also being created downstream.

Options to slow and store flow can be delivered in headwaters, but proof that they remain fully operational and effective in flood events is still limited, as is how much benefit is accrued from the investment.

4.4 Run-off pathway management

4.4.1 Introduction

This section describes run-off pathway management measures that can be added to farmed landscapes to slow and store flood flow, and consequently increase travel time. Although this literature review refers to these measures as run-off pathway management measures, they are most often referred to as run-off attenuation features (RAFs), rural SuDS (RSuDS) and Natural Water Retention Measures (NWRM). Table 4.4 presents a summary of the terminology used for run-off pathway management measures.

RAFs, RSuDS and NWRM encompass a number of other measures (that is, ditches, buffer strips and woodland) that are not discussed here. Instead this section looks exclusively at making space for water through constructed features that lie within farmed fields and other surfaces within the rural landscape, but not within drainage channels. These measures can store and disconnect both overland flow and channel flow, increasing travel time. They include:

- farm ponds (permanent ponds and dry retention ponds)
- sediment traps
- fences
- swales

This section uses the terms RAF and RSuDS interchangeably depending on the terminology used in the references cited.

The numerous terms used for run-off pathway management measures have similar definitions and functions to intercept and attenuate hydrological flow pathways by collecting, storing and improving the quality of surface run-off water within rural catchments. Originally, many of these terms came from other purposes such as farm pollution issues, but included an element of flood risk. Table 4.4 outlines the multiple definitions.

Terminology	RAFs	RSuDS	NWRM
Definition	A man-made landscape intervention that intercepts and attenuates a hydrological flow pathway to provide multiple benefits, including flood management and improving water quality (Quinn et al. 2013)	Comprise [of] individual or multiple linked component structures replicating natural processes, designed to attenuate water flow by collecting, storing and improving the quality of run-off water within rural catchments. The simple definition is that they are measures that primarily intercept run- off or drainage pathways (Environment Agency 2012)	Multifunctional measures that aim to protect water resources and address water-related challenges by restoring or maintaining ecosystems as well as natural features and characteristics of water bodies using natural means and processes (NWRM 2015a)

Table 4 4	Terminology used for run-off nathway management measures
1 avic 4.4	reminology used for run-on pathway management measures

Terminology	RAFs	RSuDS	NWRM
Relevant measures in this section	Ponds	Ponds, sediment traps, swales	Ponds, swales
Other related measures	Bunds, drain barriers, run-off storage features (online and offline), LWD dams, buffer strip management and willow barriers	In-ditch options, woodlands, buffer strip/headland options, wetlands, farm buildings	55 NWRM are listed, including urban measures

The measures described in Sections 4.2 and 4.3 build on the intrinsic storage capacity of the soil to suppress fast flow pathways and improve the function of the local drainage channels. Sections 4.1 to 4.3 focus on managed landscapes, especially intensively farmed landscapes that have lost their natural attenuation capacity; the landscape has been altered so that flow pathways now have a greatly increased connectivity and conveyance rate. Run-off pathway management measures, such as farm ponds or swales, seek to redress this imbalance. It is possible, in principle, to add high levels of attenuation capacity to a catchment if an extensive network of structures is installed. Restoring storage capacity and slowing flow rates at the catchment scale is key to NFM implementation. The evidence behind the role of these features and an assessment of their contribution to catchment-scale flooding is reviewed and discussed below.

4.4.2 Understanding the science

Run-off pathway management measures are intended to mimic natural hydrological regimes to minimise the impact of human activity on surface water drainage discharge, reducing flooding and pollution of waterways and groundwater (Environment Agency 2012). They have the potential to regulate run-off through the temporary storage of floodwater, disconnection and lengthening of flow pathways, or increasing travel time, and roughening the floodplain during flood events (Nicholson et al. 2012). They provide structural measures, primarily to control surface run-off, by helping to buffer peak flows and thereby contributing to flood risk management (Environment Agency 2012).

The evidence that individual run-off pathway management measures operate efficiently during the peak of storms is uncertain. However, the Environment Agency's RSuDS Manual and the 'catchment-based flood solution' seen at Belford and the Eden DTC emphasise the importance of bringing together a suite of constructed features that can work together as part of the 'treatment train'; this is more effective than individual measures (Environment Agency 2012). The RAF approach (Quinn et al. 2013) advocates the use of many features located throughout the landscape, with the benefits accrued by the network of features rather than one large scale/dominant intervention (Nicholson, et al. 2012). The approach does not seek to replace more traditional flood management options, but rather to add to the number of potential options available to flood risk managers (Quinn et al. 2013). This is a similar conclusion to that of the Eye Brook work suggested by Water Friendly Farming and The Flow Partnership (Biggs et al. 2016). Modelling indicated that installing more interception features, such as 'permeable dams' which hold back floodwater for a few days allowing it to drain slowly, could reduce the 1 in 100 year flood peak by 20% (Biggs et al. 2016).

This section of the literature review delves further into specific run-off pathway management measures including farm ponds (both permanent ponds and dry retention ponds), sediment traps and swales, which are recommended to be used in combination.

Farm ponds

Farm ponds are identified in the European Commission's NWRM Synthesis Document No. 1 (NWRM 2015a) and the RSuDS Manual (Environment Agency 2012) as detention, infiltration and retention ponds (Table 4.5).

Rural SuDS component	Description	Reference in this literature review
Detention	Normally dry basins designed to temporarily store and slowly release run-off water through outfalls	Dry retention ponds
Infiltration	A depression designed to store run-off and infiltrate into the ground; can include an overflow pipe	Dry retention ponds
Retention	Wet ponds designed to permanently retain some water at all times and provide temporary storage above it, through an allowance for large variations in level during storms	Permanent ponds

Table 4.5Description of farm ponds

Source: Environment Agency (2012a), NWRM (2015c)

Farm ponds can be a type of water retention structure that add flood retention capacity as either a permanent farm pond (a wet pond) or a temporary pond that is designed to dry out over time (a dry retention pond). Farm ponds are not new concept. A healthy commercial business exists for pond construction for numerous reasons, whether it is for stock water supply or crop irrigation. The UK Irrigation Association provides numerous best practice guidance examples through its series of irrigation booklets on topics including drought protection and water quality.²⁹ However, there is little design motivation for farm ponds to create flood storage as many ponds are lined and are targeted at irrigation optimisation. Therefore, there is a need to define the purpose for which the pond is being created and whether it can be modified to hold flood flow, or have spare capacity during large events.

Permanent ponds

Permanent ponds are designed to permanently retain water at all times and can increase flood retention capacity through temporary additional storage above the normal pond operating level, which allows for large variations in levels during storms (Environment Agency 2012). They differ from constructed wetlands by having a greater average depth of water. Run-off from rainfall events can be slowed and temporarily stored within the pond, and can encourage groundwater recharge, promote pollutant removal and increase biodiversity benefits (Environment Agency 2012). In this context, permanent ponds operate within fields and target overland flow paths and channel flow.

There is little literature known that identifies how permanent ponds can increase flood storage in the landscape or increase travel time to surface water bodies. Studies such as Verstraeten and Posen (1999) assumed this function to reduce run-off without providing supporting quantitative evidence. Verstraeten and Posen (1999) stated an initial assumption that retention ponds 'hold up the storm run-off for a certain time and so limit the peak discharge to a level that is convenient for the drainage system'. Their study also noted that

²⁹ <u>http://www.ukia.org/irrigationbooklets</u>

there was an increase in retention ponds being built in Belgium for the purpose of storing direct run-off from arable land (Verstraeten and Posen 1999).

Alongside the extremely limited qualitative evidence on reducing flood risk from constructing farm storage ponds, there are limited results on modelling the impact of farm storage ponds. A modelling exercise by Heathwaite et al. (2005) found that small ponds which store overland flow temporarily at the bottom of a field were very effective in reducing overland flow following storm events. Research as part of the Parrett catchment study modelled flood retention ponds in a number of different scenarios which implied that a 1% coverage was needed for a significant (but modest) impact on surface run-off (McIntyre et al. 2012). However, in the Parrett catchment, it would be difficult to achieve such coverage in practice.

Dry retention ponds

Dry retention ponds and scrapes (shallow depressions with gently sloping edges), hold water seasonally and can dry out depending on the climatic conditions (Environment Agency 2012). The primary function of a dry pond zone is to hold back large amounts of ephemeral, overland flow for a few hours or days. Given the nature of how pond zones may fill and empty, some fraction of the pond may remain wet. Water can leave the basin via a restricted outflow control, leading to a longer detention time and improved sedimentation (Environment Agency 2012). The RSuDS section below discusses how the build-up of sediment may occur where the water ponds for longest. However, the occurrence of sediment trapping and the creation of wetter zones have been encouraged as a good design aspect by Water Friendly Farming (Biggs et al. 2014).

An important component of the construction of dry retention ponds as a feature to mitigate flood risk is the controlled outflow either from a constructed outlet or percolation through the leaky nature of the RAF. Leaky bunds or barriers allow for ponding of the peak flow and the type of soil can allow for some infiltration. They can be constructed in the form of soil bunds, such as seen in the Nafferton Farm Research Project and Belford, or constructed materials such as the permeable timber barriers found at Belford.

Researchers in the Belford catchment (~6km²) installed a series of RAFs as part of the Farm Integrated Run-off Management (FIRM) plans. The RAFs were initially constructed to demonstrate to farmers the concept of RAFs, illustrating their ability to store run-off from both overland flow and channel flow diverted from the stream (Wilkinson et al. 2010a). The pilot RAF, which was constructed using permeable timber barriers, diverted peak flow from the stream using a control structure in the form of a V-notch weir and stored it during a storm event. During high magnitude storm events, the flow diverted into the piloted RAF from the stream can be as much as 15% (Nicholson 2014). The run-off was successfully attenuated within the RAF for ~8 hours (Wilkinson et al. 2010b). The slow drainage of the floodwater through the timber permeable barriers increases the travel time and allows it to continue moving through the catchment; the pond is designed to completely empty before a subsequent flood event (Nicholson 2014). The results of the pilot RAF in the Belford catchment indicated that the pilot RAF increased the travel time of the peak from 20 to 35 minutes compared with the peak flows before construction (Wilkinson et al. 2010b). The storage capacity of the RAFs was small, but the attenuation effects of the features on the flood hydrograph could be seen (Wilkinson et al. 2010b).

The Nafferton Farm Research Project (Quinn et al. 2007a, Environment Agency 2012) investigated a number of different measures (including RAFs) that worked to:

- reduce pollution
- store and slow run-off
- trap and recycle waste on the farm

Figure 4.30 shows 2 examples on ponds at Nafferton Farm.



Figure 4.30 Corner of field ponds at Nafferton Farm for local run-off from the field and the farm tracks

Source: Quinn et al. (2007a)

The research showed that ponds, barriers and bunds can physically store large amounts of run-off, helping to slow flow by creating 'transient storage' (Quinn et al. 2007a), but did not include any supporting quantitative evidence.

Similar techniques were tested in the Zwettl/Kamp catchment in Austria as part of the CRUE project when microponds were used effectively to manage hillslope run-off (CRUE 2008). The structure of these microponds is slightly different to dry retention ponds because they lack an outlet flow (provided either by a pipe or through the structure). Instead, they drain very slowly via percolation into the soil. This method separates the fast run-off stored in the ponds from the rest of the water in the storm hydrograph due to the slow groundwater processes taking place. The lack of outflow, however, means that the micropond would be ineffective if 2 extreme rainfall events were separated by, for example, one day. The microponds discussed in the CRUE (2008) report have an average storage capacity of 100m³, which means there have to be (and are) thousands located throughout the Zwettl/Kamp catchment in order to have any impact on attenuating hillslope run-off.

Sediment traps

Sediment traps are found in many different forms, but typically consist of an excavation located on a surface run-off pathway (SEPA 2015). The run-off enters the excavation area and is detained there, allowing sediment to settle before the run-off is discharged, usually via a gravel outlet (SEPA 2015). Ditches can also act as sediment traps. In this section, sediment traps are discussed in the context of being placed within a field and particularly in locations where flow and sediment would exit a field.

A sediment trap intercepts overland flow pathways into a containment area where sedimentladen run-off is temporarily detained under quiescent conditions, allowing sediment to settle out before the run-off is discharged (Environment Agency 2012). As they can take a number of different forms from simple constructions to more engineered constructions, other run-off management measures can essentially function like a sediment trap, particularly sedimentation boxes and farm ponds.

A simple sediment trap is designed so that the flow path is interrupted allowing particulate matter to settle out. Alternatively, drainage from a small surrounding catchment is collected in the trap, where it is allowed to settle before passing out through the outflow (Environment Agency 2012), as seen in Figure 4.31.



Figure 4.31 Diagram of a simple sediment trap

Source: Environment Agency (2012a)

The RSuDS Manual states that sediment traps can attenuate peak flows due to detention trap volume, but that once the trap is filled, there is minimal impact on flows (Environment Agency 2012). The RSuDS Manual (Environment Agency 2012) and the SEPA Handbook (2015) both stress that sediment traps are unlikely to achieve significant flooding benefits on their own but that, in conjunction with other run-off management features, they can help to control the release of sediment to the river network and thus maintain the capacity of rivers to convey floodwaters.

The Eden DTC study for the Morland mitigation subcatchment was designed to show the potential impact of a treatment train of flow interception options (Barber et al. 2016). The treatment train consisted of 7 mitigation features and farmyard interventions which included sediment traps, farm ponds, ditches and cross drains. It was applied to a 1.6km² mitigation study catchment as a holistic solution to help alleviate diffuse water pollution from agriculture. During numerous storms, the study site exhibited large overland flow pathways that would propagate for long distances across the farm (Owen et al. 2012). These known pathways were targeted and intercepted by temporary storage zones designed primarily as sediment traps. By disrupting and attenuating the overland flow, the time taken for the water to reach the channel was increased, potentially reducing the flood peak (Wilkinson et al. 2010b, Owen et al. 2012). Although this claim was not supported by any data or evidence, it could be further analysed in the future as an automatic weather station has been installed in each of the Eden DTC's 3 focus areas. The automatic weather stations can log rainfall, air temperature, radiation, wind speed and wind direction every 15 minutes, and include a water level recorder (Owen et al. 2012).

Sediment fences

A sediment fence is a special type of sediment trap, which also doubles as a leaky barrier. Sediment fences are referred in the RSuDS guide published by CREW as an 'other management option' that can be used to mitigate diffuse pollution from steading and field run-off (Duffy et al. 2016). They are:

'constructed along the lateral slope of fields and comprise of a narrow weave geotextile mounted on wooden posts, with the geotextile anchored (buried) beneath the soil surface' (Duffy et al. 2016).

The fence intercepts field run-off, trapping soil and allowing water to percolate through the geotextile. According to the CREW guide, they are typically used on moderate slopes for high risk crops (for example, potatoes) where there is a flow with a high sediment content (Duffy et al. 2016).
There is currently no quantitative evidence of sediment traps reducing flood risk. However, the principles of how they work are similar to those of other run-off pathway management measures in increasing the travel time and potentially increasing the flood storage capacity.

Sediment fences have been trialled in the Lunan catchment in eastern Scotland after the harvest of potatoes in an attempt to prevent soil erosion and diffuse pollution (Vinten et al. 2014). Evidence of their benefits were found with regard to mitigating diffuse pollution but not within the context of reducing flood risk, as the increase in travel time was not quantified. More data need to be collected.

Swales

A swale is defined by the CREW RSuDS guide as 'a linear, dry, grass channel laid with a shallow fall on its base'. A grassed waterway is another term used for swales, as seen in the RSuDS Manual (Environment Agency 2012).

Swales are designed to collect and transfer run-off during rainfall events. Swales are dry channels and water is normally only found in them following rainfall (Duffy et al. 2016). Swales in the context of farm steadings are not discussed in this literature review but further information can be found in the CREW RSuDS guide (Duffy et al. 2016).

Swales are wider than 10m, larger than grassed buffer strips along field borders (up to 6m wide and 200m long), and have a potential to reduce run-off volume and peak discharge rate, especially in small catchments up to 15ha (Fiener and Auerswald 2006). Figure 4.32 illustrates a typical swale design.

What does a Swale look like?



Figure 4.32 Typical swale design. Left: a well-maintained feature and (right) a less maintained, roughened surface and containing a stone check dam

Source: Duffy et al. (2016)

In-field swales are typically part of the treatment train used to transfer run-off between 2 RSuDS or from a RSuDS to the end of a treatment train (that is, from a sediment trap bund into a farm pond or river) (Duffy et al. 2016). Swales are cheaper to construct than piped systems. They form a complementary structure to a bund, as they are excavated into the soil. Thus, water is deflected and contained within the swale depending on its size or depth. The swale can contain check dams to slow and pond further. The feature can alter the connectivity of the flow (Duffy et al. 2016), but the evidence discussed below shows that they clean, slow and potentially reduce flow.

Grassed waterways with large hydraulic roughness can exhibit considerable potential in reducing run-off, sediments and pollutants from agricultural watersheds (Fiener and Auerswald 2003). The increasing infiltration and evaporation from the grassed waterways ((Charlesworth et al. 2003), decreases connectivity and increases the travel time to surface water bodies.

The majority of the literature regarding swales originates from the urban context, such as the CIRIA SuDS Manual (CIRIA 2015) and these systems can be found internationally. This literature review focuses on evidence developed within a rural context.

4.4.3 Confidence in the science

There are few peer-reviewed studies on the installation of run-off pathway management measures in rural catchments. As seen in Section 4.4.2, the peer-reviewed studies with measured evidence that are most often cited are the Belford study and the Eden DTC in Cumbria (see, for example, Nicholson et al. 2012, Owen et al. 2012, Quinn et al. 2013, McGonigle et al. 2014). Modelled evidence can be found at a local scale in Heathwaite et al. (2005) and local study effects were upscaled to look at a catchment scale by Evrard et al. (2007) and Quinn et al. (2008a, 2008b, 2013). However, there is extensive grey literature on run-off pathway management measures, including guidance for practitioners developed by the Environment Agency (Environment Agency 2012), CREW (Duffy et al. 2016), Newcastle University and Environment Agency (2011) and the European Commission's NWRM individual measure guides (listed in NWRM 2015a).

The limited peer-reviewed measured evidence and qualitative grey literature evidence gives limited confidence to the local scale scientific evidence of the impact of implementing run-off pathway management measures to reduce flood risk. The UK's reform of rural and agricultural policy (Rural Strategy 2004) (Defra 2004b) and new strategic assessment of policy for flood risk management (Making Space for Water) (Defra 2004a) have had a major influence on the approaches to flood risk management in rural areas. Incentives are leading farmers away from intensive farming towards environmental protection and enhancement, such as diffuse pollution reduction under the Water Framework Directive (Posthumus et al. 2008). Making Space for Water (Defra 2004a) was partly in response to the need for changes in order to achieve Water Framework Directive requirements and partly in response to the widespread flood events in 2000. The management of flood risk was introduced to rural land management by installing measures to control run-off from farmland and retaining water on farmland in the higher parts of catchments, as well as storing it on floodplains in the lower parts (Posthumus et al. 2008). These policy changes have given rise to more research and grey literature with measured evidence on the effectiveness and performance of run-off pathway management measures on flood risk, but have yet to have similar presence within peer-reviewed literature.

Farm ponds

An important part of implementing a new method of rural land management for managing flood risk involves consensus from various stakeholders to make programmes such as Making Space for Water successful (Posthumus et al. 2008). The research by Posthumus et al. (2008) explored the perception of local stakeholders on the implementation of farm ponds in the Laver and Skell catchments in north Yorkshire. In 2006, a stakeholder workshop for the Ripon Multi-Objective Pilot (Ripon-MOP) project revealed that solutions such as retention ponds were suggested because it was thought they would affect surface flow connectivity by trapping, retaining or slowing down overland flow (Posthumus et al. 2008).

The modelling results obtained by Heathwaite et al. (2005) showed that small ponds which store overland flow temporarily at the bottom of a field were very effective in reducing

overland flow following storm events. However, the effect of drainage on the peak flow and flood generation still requires further research (Posthumus et al. 2008).

A number of the RAFs constructed as part of the Belford study showed qualitative and quantitative evidence of dry retention ponds reducing flood risk. RAF-11 in the Belford catchment study was a dry retention pond constructed using locally sourced soil and boulders to form a bund over the natural gully in the field. It is situated in a prominent flow path with a potential volume of 500m³ in a contributing catchment of 0.3km² (Nicholson, 2014). Data from RAF-11 (Figure 4.33) show a targeted response (as designed) to a short, intense rainfall event. The event that occurred in June 2012 caused no flooding to Belford and saw 28mm of rainfall in 12 hours, with RAF-11 achieving a water level of approximately half of its maximum. However, the field leading up to the RAF was covered with a mature wheat crop, which may have reduced the run-off in this particular event. If the field had been bare, as it has been in winter months, the surface run-off may have been greater in magnitude (Nicholson 2014). The results demonstrate a significant reduction in peak overland flow (>50%) generated in the small contributing area preceding the RAF (Nicholson 2014). This evidence, as well as further evidence presented in Nicholson (2014) on overland flow features, indicates a high local impact targeting overland flow.





Source: Nicholson (2014)

The Belford study is the only one known to provide qualitative and quantitative peerreviewed evidence of the effect of farm ponds on flood risk. The further evidence described is obtained from the grey literature. Although the science is sound, there is limited measured and monitored evidence supporting a reduction in flood risk by farm ponds.

Sediment traps

The confidence in sediment traps, fences and bunds reducing flood flows is extremely limited. Theoretically, the sediment traps should act as another form of flood attenuation feature, similar to farm ponds but on a smaller scale. However, no peer-reviewed evidence

was found which supports the claims that sediment traps can attenuate peak flows due to the detention trap volume (Environment Agency 2012).

Swales

Swales in a rural context have limited peer-reviewed evidence in the UK, but information from international studies can be found in Fiener and Auerswald (2006) and Evrard et al. (2007).

The study by Fiener and Auerswald (2006) in Munich in Germany provided qualitative evidence which identified a reduction in run-off with the establishment of a grassed waterway, and improved soil and water conservation. The study noted that the main parameter controlling the run-off reduction in the grassed waterway was the inflow, which depended on the precipitation characteristics and the physical characteristics of the management in the watershed draining into the grassed waterway (Fiener and Auerswald 2006). Further information on this study can be found in Section 4.4.5.

Evrard et al. (2007) provided modelled evidence of the effectiveness of 12ha of grassed waterways in a small agricultural catchment in Belgium to reduce flooding. The results in a worse-case scenario (December with the highest largest potential of run-off generation) showed a reduction in the peak discharge and total run-off volume by 50% (0.5m³s⁻¹) instead of 1.0m³s⁻¹) and 40% respectively (2,651m³ instead of 4,586m³), while the lag time increased by 16% (Evrard et al. 2007). The study considered 2 reasons for a reduction in the generated run-off volume for the worse-case scenario:

- reduction in run-off velocity in the grassed waterways over sparsely covered cropland
- lower run-off coefficient used since less run-off would be produced in the grassed waterway itself and the run-off velocity was also reduced in the grassed waterway

These studies provide confidence in the ability of swales to reduce flood risk through a reduction in total run-off through infiltration and evaporation, and their ability to increase travel time. However, it should be noted that the effect of swales on water flows is dependent on local conditions as described by Fiener and Auerswald (2006) and the delay in storm water run-off peak may need to be further examined for a synchronicity issue (see Section 4.4.1).

Catchment scale

Although the results of Quinn et al. (2013) for the Belford Case Study in Northumberland (~5km² catchment) show that individual small-scale RAF storage measures contribute to flood attenuation to a small extent, the authors acknowledged that the effectiveness for flood risk management lies in understanding how they integrate into the hydrological response of the entire catchment. A more pertinent issue is the confidence in the operation of a cluster of measures and the performance at the larger catchment scale and in flood level events (Quinn et al. 2013). The modelling of RAFs is limited, as there is no simple methodology or tool for designing an appropriate network of RAFs to achieve a desired level of reduction in flood hazard for a particular catchment (Quinn et al. 2013). Catchment-scale modelling simulations were performed for the Belford catchment using a simple lumped conceptual rainfall–run-off model (TOPMODEL) (Quinn et al. 2008a). A calibrated model was used to simulate the storage/attenuation effects of the RAF network in Belford using a simple routing function (a unit hydrograph), which was calibrated to emulated the output of the Pond Network Model (Quinn et al. 2013). The study concluded that a simple hydrological model

had the potential to emulate the impact of a RAF network on flow (mm per hour) for small catchment impact studies (<10km²), although it was recognised there were still uncertainties in the predictions (Quinn et al. 2013).

A catchment-scale modelling simulation of land use management was conducted for Pontbren in mid Wales to demonstrate the effects of storage ponds along with shelter belts, buffer strips, and improved and unimproved grassland (Wheater et al. 2008). The results indicated that careful placement of such interventions can significantly reduce the magnitude of peak run-off at the field and small catchment scale. This evidence indicates that permanent ponds, in combination with other run-off management measures, can significantly reduce the magnitude of peak run-off at the field and small catchment scale. The methodology developed by Wheater et al. (2008) has the potential to represent and quantify the catchment-scale effects of upland management. However, literature covering the impact of farm ponds, sediment traps and swales is limited. It is physically reasonable that the addition of new storage to a catchment should help reduce run-off rates and catchment models can be constructed to demonstrate potential effects (Nicholson 2014).

Further evidence is being collected to determine the impact of run-off pathway management measure networks at a catchment scale. The Eden DTC, Wensum DTC and Hampshire Avon DTC are 3 study areas containing various mitigation features and farmyard interventions (that is, sediment traps, farm ponds and so on) designed primarily to gather empirical measured evidence on the cost-effectiveness of combinations of diffuse pollution mitigation measures at catchment scales (Owen et al. 2012). The evidence of the impact of agricultural diffuse pollution can be found in studies by Owen et al. (2012) and McGonigle et al. (2014); however, there is limited evidence on the impact of the combination of measures on reducing flood risk. The Eden DTC monitoring results showed that the attenuation of overland flow in sediment traps increased the time taken for water to reach the channel and could potentially reduce the flood peak (Owen et al. 2012).

Lastly, there are many reservations about how well storage features operate in very large events, that is, when the capacity is full. There is little evidence about:

- what determines how many structures or the extent of the treatment train are necessary to make a difference to large flood events
- if there are any negative impacts of local management when it comes to flow synchronisation at the larger scale

Hence, the bulk of the evidence described here has more focused on stressing that:

- a well-designed feature can operate as designed in the field
- there is potential to use such features part of the NFM toolkit

However, more research is needed on scaling up issues.

4.4.4 Metrics of cost–benefit and multiple benefits

RSuDS, RAFs and NWRMs are often defined by their multiple benefits which include attenuating water and improving the quality of run-off water by slowing, storing and filtering run-off water (Newcastle University and Environment Agency 2011, Environment Agency 2012). Run-off pathway management measures consequently have several additional purposes such as sediment trapping, prevention of downstream linear erosion, water quality management, carbon sequestration and biodiversity (Fiener et al. 2005, Owen et al. 2012).

A number of manuals and handbooks review run-off pathway management measures, with some including a determination of the associated multiple benefits. They include:

- 'Rural Sustainable Drainage Systems: A Practical Design and Build Guide for Scotland's Farmers and Landowners' (Duffy et al. 2016)
- 'Rural Sustainable Drainage Systems' (Environment Agency 2012)
- 'Run-off Attenuation Features: A Guide for all those Working in Catchment Management' (Newcastle University and Environment Agency 2011)
- 'Individual NWRM: Retention Ponds' (NWRM 2015c)
- 'Individual NWRM: Swales' (NWRM 2015d)

The RSuDS Manual (Environment Agency 2012) also offers details of the performance and cost measures of sediment traps, swales and ponds. These manuals and handbooks do not provide peer-reviewed evaluations of the ecosystem services and benefits but are intended as practical guidance.

Farm ponds

Numerous peer-reviewed works have reported on the use of constructed wetlands and related farm pond systems for reducing farm-based and urban pollution loadings (see, for example, those listed in Kay et al. 2010).

The Belford catchment study (5.7km²) installed 2 RAFs to reduce concentrations of suspended sediment (SS), phosphorus (P) and nitrate (NO₃) in run-off (Barber and Quinn 2012). These measures were constructed principally for flood attenuation purposes, but designed to include water quality benefits to determine if RAFs could have multiple benefits. Although the field bund RAF had the ability to retain significant volumes of sediment, the underlying field drains still exported high concentrations of sediments and nutrients, occasionally exceeding 500mg SS I⁻¹, 1mg TP I⁻¹, and 40mg NO₃ I⁻¹ (Barber and Quinn 2012). The residence time requirements to reduce suspended sediment, nitrate and phosphorus loadings during peak flows needs to considered when installing RAFs and wetlands; a catchment is likely to require either very large features or large numbers of small ones (Barber and Quinn 2012).

Mitigation Feature 3 in the Eden DTC is a retention pond that intercepts and temporarily stores track run-off (50m³) from a 15ha contributing area with annual accumulations of 87kg of sediment per hectare, 0.1kg of total phosphorus per hectare and 0.3 kg of total nitrogen per hectare in 2014 to 2015 (Barber et al. 2016). Other farm pond features in the Eden DTC were found to capture pollutant loads in the mitigation features covered by the study (Barber et al. 2016).

The MOPS project also constructed a number of run-off pathway management features in fields and on field edges for reducing pollution from arable fields (Deasy et al. 2010). The initial results from the retention ponds (known as constructed wetlands in this project) suggest that the bare soil and muddy ponds during construction act as a sediment and nutrient source rather than a sink until the wetland vegetates naturally. The sediment in the retention ponds is to be further analysed to assess the fertiliser value of the dredged sediment for farmers (Deasy et al. 2010).

Lastly, the well-documented study on retention ponds in Belgium by Verstraeten and Poesen (1999, 2000) provided evidence of dry retention ponds storing large quantities of sediment from run-off events. The data from the dredged sediment volumes were able to provide a predicted sediment yield for the drainage basins, with values varying between 0.19 and 6m³ per hectare per year for basins ranging from 25 to 5,000 hectares (Verstraeten and Poesen 1999).

Sediment traps

As previously discussed, sediment traps were initially constructed to trap sediment by intercepting overland flow pathways and temporarily detaining the flow to allow sediment to settle out before the run-off is discharged (Environment Agency 2012). Consequently, the multiple benefits of sediment traps effect on pollution trapping are found in studies featuring sediment traps.

Mitigation Feature 1 in the Eden DTC are sediment traps that in 2014 to 2015 had annual accumulations of 263kg of sediment per hectare, 1.2kg of total phosphorus per hectare and 2.9kg of total nitrogen per hectare with a storage volume of 100m³ (total of 2 sediment traps) from a contributing area of 1.9ha (Barber et al. 2016).

Swales

As previously stated, a large majority of the literature regarding swales originate from SuDS. The review of SuDS by Charlesworth et al. (2003) examined swales and determined that grass swales removed particulate-associated contaminants by sedimentation, filtration through the grass lining of the swale and adsorption onto soil particles upon infiltration Ellis (1992) found that swale 30–60m in length could retain 60–70% of suspended solids and 30–49% of metals, hydrocarbons and bacteria.

Evidence provided in a rural context is limited. The study by Fiener and Auerswald (2006) found that grassed waterways without maintenance for 9 years exhibited considerable potential in reducing sediment delivery, as well as run-off, from an agricultural watershed. The grassed waterways in the experiment's 2 watersheds reduced sediment delivery by 90% and 100% respectively (Fiener and Auerswald 2003). The study by Evrard et al. (2008) in the Belgian loess belt of a 12ha grassed waterway and 3 earth dams showed sediment discharge reductions of 93% between the grassed waterway's inflow and the outlet. The specific sediment yield was reduced and sediment transfer decreased dramatically, reducing the damage costs associated with muddy floods in the study area (Evrard et al. 2008).

Upscaling

Although benefits are often shown at a local field or ditch scale, there is still little evidence that these features work to control pollution or protect freshwater life at a scale of whole streams or rivers (Stoate and Biggs 2016). The Water Friendly Farming study and the DTC studies are looking to fill this gap. The Water Friendly Farming study seeks to provide data to determine to what extent introducing landscape-wide mitigation measures can reduce rural water pollution, hold back floodwater, and protect freshwater biodiversity (Biggs and Stoate 2016). The results from the Water Friendly Farming study showed that installing edge of field measures to control run-off from the clay-dominated landscape has modest effects in the 3 years studied (2011 to 2013), with the continuous monitoring of water quality at the downstream end of each catchment showing that phosphorus levels had increased, nitrogen levels had fallen slightly and sediment levels had not shown a consistent pattern (Biggs and Stoate 2016). However, the Eden DTC results showed a total reduction in pollutant loads from 5 measured mitigation features of 5.3 tonnes of sediment, 9.7kg of total phosphorus and 17.8kg of total nitrogen, equating to specific yield reductions of 42kg of sediment per hectare, 0.06kg of total phosphorus per hectare and 0.16kg of total nitrogen per hectare at a 1.6km² catchment outlet (Barber et al. 2016).

The Water Friendly Farming study is still seeking to collect empirical evidence to test the hypothesis that adding new clean water ponds to the landscape increases landscape-scale freshwater biodiversity (Biggs and Stoate 2016). Current data available for landscape-wide freshwater biodiversity are measured in terms of wetland and aquatic plants. In the Stonton

Brook catchment, new ponds were rapidly colonised and the submerged aquatic plants led to a consistent and landscape-wide increase in freshwater biodiversity (Biggs and Stoate 2016). Biggs and Stoate (2016) concluded that further monitoring was needed to determine whether this increase was permanent.

Cost-benefits

There is limited peer-reviewed evidence of the cost-benefits of run-off pathway management measures. Verstraeten and Poesen (1999) performed conducted an economic evaluation of small-scale flooding, muddy floods, retention pond construction and sedimentation. The study looked at direct costs relating to the flooding process itself, and indirect costs relating to control measures emphasising the difficulties of determining what should be included (that is, the intervention of the fire department that does not charge for its work), valuing nonmaterial items (that is, working hour) and even collecting the necessary information (Verstraeten and Poesen 1999). The total estimated damage to households in 3 drainage basins in southern Limburg in central Belgium during 1973 to 1998 amounted to ~€2.7 million (1999 figure). The indirect costs included the construction of a retention pond. The mean cost of more than 100 retention ponds constructed for the study was €380,000, with an annual maintenance cost of €1.5 million for regular dredging. The costs could increase where sewage water from households or industry join the run-off before entering the retention pond as the sediment becomes polluted with heavy metals, phosphates, fluorides and chlorides, and has to be evacuated. Further costs and breakdowns can be found in Verstraeten and Poesen (1999).

Quinn et al. (2007a) contained a critique of costs and implementation of FIRM plans, which included run-off pathway management measures. The cost of some of the features shown below reflects the full estimated cost incurred by the project:

- construction of infiltration pond £7,000 each
- construction of 5m concrete section in sediment trap £1,000 each
- Ochre P trap £2,000

Table 4.6 Estimate of physical storage achieved and costs at Nafferton Farm

Feature	Cost (£)	Storage (m ³)	Rainfall depth (mm)	Cost (£) per mm stored
Ponds	14,000	250	0.25	5,600
Sediment trap	1,000	25	0.025	40,000
Barriers (when full) ×4	4,000	100	0.1	40,000
Wetlands 60m × 3m	11,000	5	0.005	2,200,000
Total	30,000	380	0.38	78,947
Total without wetlands	10,000	375	0.375	50,666

Source: Quinn et al. (2007a)

At Nafferton Farm, only ponds were able to provide a reasonable amount of storage in a single feature. The other features were placed into ditches for multiple reasons and therefore the disproportionate cost of some features needs to be balanced against the other functions they perform (for example, sediment trapping) (Quinn et al. 2007a). It is worth noting that the wooden structures were more expensive than the soil bunds. Typically, a wooden bund of 400m length would cost £8,000 and a single soil bund with a similar capacity would cost

 $\pounds 2,000$. The costs of the Nafferton Farm interventions are costly, but the study on flood storage on farms noted a number of considerations that needed to be taken into account. For example, costs would be lower if installed by local farmers and local agricultural engineers; measures were currently overdesigned for research needs and benefits needed to be taken into account (Quinn et al. 2007a).

The study by Quinn et al. (2007b) on nutrient pollution proposed a FIRM plan for a theoretical farm similar to Nafferton built on its findings and those of Quinn et al. (2007a) to address both the nutrient pollution and flood storage problems. The theoretical catchment is a 1km² square catchment on a typical farm with a field drain, 6 fields and 500m of ditch/channel, buffer strips with bunds and fencing (or hedgerow). Further details on this theoretical farm and its costs of measures over the life cycle of the FIRM plan can be found in Quinn et al. (2007b). It was calculated that it would cost ~£37,000 over a 5-year FIRM plan or £7,400 per km² per year, which could be calculated as £6,896 per km² per mm of run-off stored, where no other farmland is inundated (Quinn et al. 2007b). These analyses are estimates; they do not include the effects of transient storage and are subject to great uncertainty (Quinn et al. 2007b).

Lastly, a number of these features such as sediment traps and ponds are eligible for a capital grant in areas targeted for the reduction of water pollution from agriculture. These schemes are also considered 'payment for ecosystem services' or agri-environmental schemes. This literature review does not review these schemes extensively, but some references include Morris and Potter (1995) and Kleijn et al. (2006).

4.4.5 Effectiveness/performance

Biggs et al. (2016), Verstraeten and Posen (1999), Heathwaite et al. (2005), Evrard et al. (2007), Quinn et al. (2007a, 2007b) and others have shown that run-off pathway management measures have a positive flood risk management benefit especially at source, within hours of the flow being generated (Quinn et al. 2007a). The previous sections have highlighted the performance of these measures.

The Nafferton Farm study found that if a typical farm or small catchment can sacrifice 2–10% of the landscape to run-off storage and mitigation features, the properties of the run-off regime should be significantly altered.

To maximise effectiveness of run-off pathway management measures, they should be located in areas of high surface connectivity or areas where the river and floodplain are able to interact (Nicholson et al. 2012). Further discussion of how to site RAFs using GIS systems and LiDAR (light detection and ranging) are discussed in Section 4.4.7 section. It has been argued that on-farm interventions such as run-off pathway management features in the corner of fields offer greater flexibility and reversibility than traditional engineered flood defences (Deasy et al. 2010). However, field corners may have multiple drain outfalls which can cause problems and necessitate drainage diversion (Deasy et al. 2010).

Another factor that often has an impact on the effectiveness or performance and maintenance of run-off pathway management measures is sediment removal (Verstraeten and Posen 1999). This factor is further discussed in Section 4.4.7.

At the catchment scale, Wilkinson and Quinn (2010) stated that to understand the effectiveness of the mitigation measures, it is important to characterise the catchment before, during and after change. A long period of background data before installation of NFM measures would be ideal.

Farm ponds

The research shows some evidence of the effectiveness and performance of farm ponds in reducing flood risk management. Quinn et al. (2007a) found that to increase the effectiveness of ponds during storms to reduce flood risk, much larger ponds or larger numbers of ponds would be required to store the flow from a storm. To determine the size/capacity of the ponds installed for effectiveness and performance depends on the amount of rainfall in a typical storm, or an analysis of historical data to determine the rainfall conditions that caused flooding (Quinn et al. 2007a).

Based on historical data of storm events in Belford, it was determined that streams begin to flood at a magnitude above 3.5mm per hour and consequently would require storage over a 4-hour period for 1mm (Nicholson et al. 2012). Scaled up, ~20,000m³ of storage would be required to be an effective form of flood risk management (for the smallest floods) for a catchment the size of Belford (Nicholson et al. 2012). The pilot RAF, an attenuation feature with permeable timber barriers, disconnects and attenuates run-off from major pathways and holds water from the stream when it is in flood (Wilkinson and Quinn 2010). The RAF holds ~800m³ of water and will take ~8–12 hours to drain from full to empty; the effectiveness of this attenuation process can be seen in the stream flow characteristics (Wilkinson and Quinn 2010). Before the installation of the RAFs, the delay between peak flow and the diversion weir and a point 1km downstream was an average of 20 minutes (Wilkinson and Quinn 2010). Post construction, the average delay, based on 4 peaks, was 35 minutes, increasing by an average of 15 minutes (Wilkinson et al. 2010b). This RAF was seen as effective through visual observations and the data collected.

Nicholson et al. (2012) also noted that in order for the storage to be effective during the peak flow period, storage must not have been utilised prior to the arrival of the flood peak because the benefits might be much reduced. The intention of the design of the pilot RAF at Belford was for it to be completely empty before the arrival of a subsequent rainfall event. Evidence is being gathered to determine how RAFs perform during a double-peaked storm. Wilkinson et al. (2010b) presented a graph on how the pilot RAF in the Belford catchment functioned during a double-peaked storm event in September 2008 using data obtained from the instrumentation within the RAF. The data and analysis led to the modification of the pilot RAF so the water could drain slightly faster from the structure (Wilkinson et al. 2010b). No further supporting evidence was found regarding whether this modification was helpful in later double-peaked storms.



Figure 4.34Data from the pilot pond during September 2008 storm eventSource: Modified from Wilkinson et al. (2010b, Figure 5)

Evidence of more total water storage capacity in a catchment can be captured in catchment models and could link to the storage discharge patterns of flow, often best seen in the recession curve of an events. The loss peaks could also be captured in the model. This is shown more clearly in Section 2.4 (leaky dams and woody debris) and in the Belford Case Study.

To remain effective in reducing flood risk, farm ponds must be maintained regularly to ensure sedimentation remains at a low level depending on the rate of sediment influx and the size of the ponds. Survey techniques are being used to determine sediment accumulation in the MOPS project's retention ponds, which are expected will need dredging after 5–10 years, depending on the size and sediment load (Deasy et al. 2010).

Sediment traps

The effectiveness and performance of sediment traps are not well-documented with regard to reducing flood risk management. However, they do function similarly to farm ponds – though generally on a smaller scale – which allows for overland flow to be slow and sediments to settle. Factors that do have an impact on the effectiveness of sediment traps, similarly to farm ponds, is the location, size and antecedent conditions. An important feature of sediment traps that can also increase its effectiveness is the bunds or barriers surrounding them, as seen in the Water Friendly Farming Project (Biggs and Stoate 2016). This theoretically increases the roughness of the land, increasing the travel time and potentially some ponding of water, which can reduce local flood risk. However, no peer-reviewed evidence was found which support this rationale or further evidence that can determine the effectiveness or performance of sediment traps.

Swales

There is limited evidence on the effectiveness and performance of swales. However, the study by Fiener and Auerswald (2006) explored the factors, such as seasonal changes, influencing the effectiveness of grassed waterways. It looked at the relationship of overall effectiveness of grassed waterways, seasonal variation and driving parameters through measured and modelled results. It was concluded that the seasonal changes in grassed waterway properties, namely soil water content and hydraulic roughness had a minor effect (Fiener and Auerswald 2006). As previously discussed, the main parameter determining the effectiveness of grassed waterways was the inflow, which depends on the precipitation characteristics and the physical characteristics of the management in the watershed draining into the grassed waterways (Fiener and Auerswald 2006). In general, the results indicate the high potential of grassed waterways for reducing run-off and sediment delivery, especially if combined with an intensive soil and water conservation system in the draining fields. Fiener and Auerswald (2006) recommended that, for conservation planning, the least effectiveness at the end of winter should be taken into account.

Further to the evidence presented by Evrard et al. (2007) as described above, Evrard et al. (2008) reported on further measured evidence of a 12ha grassed waterway and 3 earth dams installed between 2002 and 2004 in the thalweg of a 300ha cultivated dry valley in the Belgian loess belt. The catchment was extensively and intensively monitored from 2005 to 2007, with 29 run-off events being recorded in this period (Evrard et al. 2008). The peak discharge (per hectare) was reduced by 69% between the upstream and the downstream extremities of the grassed waterway. Re-infiltration was also observed in the grassed waterway and the run-off coefficient decreased by a mean of 50% between the extremities of the grassed waterway (Evrard et al. 2008).

In addition, the SuDS Manual (CIRIA 2015) and the Susdrain website³⁰ describe good effectiveness of a well-constructed swale and bund system. Maintenance is required, which can add to costs but in principle, any roughened surface or retention zone should effectively slow and store water and trap sediment. Clearly these features are designed for urban areas and may not reflect the deluge of sediment and run-off produced by an arable field.

4.4.6 Key case studies

A number of the case studies highlighted in Section 4.4 do not look at run-off pathway management measures in isolation but as multiple measures, including other NFM measures.

Two key case studies for the UK with monitored evidence within peer-reviewed literature are the Belford catchment in Northumberland and Eden DTC in Cumbria. Although only preliminary evidence is available from these case studies, more data are to be collected.

The **Belford** catchment (5.7km²) in Northumberland provides the most extensive monitored and modelled evidence in peer-reviewed literature in the UK. The Belford Case Study consists of a number of within-field RAFs that target fast overland flow pathways and aimed at altering flood flows. The catchment study relies on the function of a network of RAFs, woody debris and offline ponds to operate together to slow and store flow (Wilkinson et al. 2010a). Many of the features were instrumented, evaluated and simulated with complex hydraulic models (Quinn et al. 2013). The research shows that the features work as designed and slow the flow. Even in larger events, the leaky structures quickly empty to allow more flood flow to be stored in the next event (Quinn et al. 2013). The measures have not affected farm productivity and some farm benefits have accrued, such the use of soil dams as roads. The Belford work suggested that a hypothetical pond network providing 19,250m³ of storage reduced peak flow by 15–30% for both observed events and FEH design storm events (Quinn et al. 2013). The Belford study has been active since November 2007. It has already yielded considerable data and will continue to further determine the impact of RAFs on flood mitigation. See the Belford Case Study for more details.

The **Eden DTC** Morland mitigation subcatchment project in Cumbria investigated the potential role of a number of local features implemented across the catchment on flooding, pollutant retention, carbon sequestration, habitat creation and biodiversity (Owen et al. 2012). These measures were measured through networks of hydrometeorological and water quality instrumentation mostly provided in real-time, with sub-hourly time steps. Along with the evidence for the benefits to nutrient pollution mitigation, simulations have also tried to address Qp reduction at the catchment scale. Owen (2016) suggested that quite large reductions in Qp and increases in time to peak are possible for the Morland site, depending on the degree of mitigation work with NFM benefits. See the Eden Case Study for more details.

There are a large number of grey literature studies ongoing or completed to determine the impact of run-off pathway management measures in the UK. Two key case studies are the Eddleston Water Project and the Water Friendly Farming work at Loddington.

The **Eddleston Water Project** in the Scottish Borders has a large network of WWNP measures, working with 20 farmers to remeander over 2km of river, to plant 200,000 native trees and to create a network of 22 ponds as well as 101 log structures (Spray 2016). The features have been instrumented and there is confidence that they are altering flow rates. The measurement of pond levels showed that the 22 ponds in the upper catchment could readily store water; however, the modelling suggested that it will only have a relatively small effect on total subcatchment run-off at this scale. Initial model results also suggest that a

³⁰ <u>http://www.susdrain.org</u>

series of larger ponds on the floodplain could reduce the discharge peak by 19–20% and delay the peak flow by up to 6 hours for a 1.5-year return interval flow event (Spray 2016)). Studies with hydraulic and catchment-scale models are ongoing and are supported by a dense network of local flow gauges.

The **Water Friendly Farming work at Loddington** in Leicestershire and in the 10km² catchments at Eye Brook and Stonton Brook are producing measurable and modelling evidence of the impact on sediment and nutrient loss and flood flow (Biggs et al. 2016). Many water quality, sediment losses and biodiversity benefits have already been reported. The study has simulated the performance of existing features and the results suggest that, when complete, a catchment-scale aggregate effect will be achieved. Modelling has indicated that by installing more of these features, particularly with the addition of 'permeable dams' which hold back floodwater for a few days in stream valleys and then allow it to drain slowly, could reduce the 1 in 100 year flood peak by 20% (Biggs et al. 2016). Further data will be collected from these measures to evaluate the potential of this technique in the lowland farmed landscape and evaluated over the next 5 years with the support of the Environment Agency.

4.4.7 Design, management and maintenance

Approach

The Wilkinson et al. (2014) study explores the catchment systems engineering (CSE) approach for agricultural management to manage flow pathways by applying it to the Belford catchment. This CSE approach is a framework to design, install, maintain and manage a treatment train, in which RAFs or soft engineered structures are at the heart of the approach (Wilkinson et al. 2014) (Figure 4.35). The test application to the Belford catchment has resulted in stakeholders using this approach to assess their own similarly scaled catchment. This CSE framework endeavours to change the flood flow regime of the catchment using an adaptive approach, including interventions with the help of local stakeholders and regulators for holistic environmental management (Wilkinson et al. 2014).

Since there are numerous ways to manage run-off pathways, the design, maintenance and management requirements will differ according to the site and its surrounding characteristics, flow pathways, financial position and other variables. There is extremely limited peer-reviewed literature on the design, maintenance and management requirements of run-off pathway management measures, but a number of guidance documents for farmers and landowners are provided by CREW (Duffy et al. 2016), the Environment Agency (2011).

Siting

The siting of run-off pathway management measures is critical since they target dominant overland flow pathways to manage flood risk. Farmers often have a great understanding of the hydrological processes within their field and this knowledge can be augmented with topographical digital data and simple flow accumulation rules to identify suitable locations (Heathwaite et al. 2005, Wilkinson and Quinn 2010). At a hillslope scale (~1km²), it is simple to identify sites where many flow pathways come together using a field visit or farmers and/or landowner's knowledge (Wilkinson and Quinn 2010). The early results of the Belford study indicated that RAFs may work best in low-order channels (Wilkinson et al. 2010b).

For larger catchments, a method of spatial analysis of detailed LiDAR or a digital terrain model (DTM) can also be applied to explore the potential siting of a number of run-off pathway management measures. The Farm Pond Location Tool (PLOT) is an example of

this method which was developed to identify ideal locations for RAFs. The Farm PLOT tool interpolates LiDAR data in ArcGIS to determine sites and cost-effective locations for disconnecting and storing run-off (Wilkinson and Quinn 2010). Furthermore, the tool can calculate a rough bund length needed based on a 0.5m or 1m high bund and the associated storage values of the feature. The flow accumulation tool within ArcView's hydrological tools can identify flow pathways to allow stakeholders to calculate the run-off contributing area that a pond may capture and disconnect on a field by field basis (Wilkinson and Quinn 2010). The outputs from Farm PLOT can be exported to Google Earth for easy accessibility and distribution to stakeholders (Wilkinson and Quinn 2010). The RAF in Belford was sited using the Farm PLOT tool (Wilkinson and Quinn 2010). Consequently, the potential site selection and storage of 500m³ could be estimated accurately and visualised for farmers, regulators and other stakeholders (Nicholson et al. 2012).



Figure 4.35 Output from Farm PLOT identifying the location of RAF-11 in the Belford catchment

Source: Wilkinson and Quinn (2010)

Design

Run-off pathway management measures have an inflow, a spillway and an outflow, similar to an offline storage area or washland. However, they usually target hydrological flow pathways, unlike offline storage areas which target watercourse levels. Targeting hydrological flow pathways often means run-off pathway management measures have an uncontrolled inflow, which means that water flows into the measure without the use of engineering solutions (Morris et al. 2004b). Swales, however, can act as an outlet pipe, or transfer run-off from one RAF to another (for example, a pond to a sediment trap), increasing the control of the inflow into a RAF (Figure 4.36). More engineered inflows (for example, sluices and adjustable weirs) are usually found in larger storage areas, such as washlands or polders; further details can be found in Section 2.5. These features require either a ponding structure such as a soil bund or a leaky barrier that causes the temporary ponding of flow. For example, larger stone dams and recycled plastic can be used to create barriers. Outflow mechanisms tend to be controlled outfalls, or infiltration into the constructed measure.

The key design attributes of run-off pathway management measures (that is, intercepting fast flow pathways) are detailed in Newcastle University and Environment Agency (2011) and Quinn et al. (2013). They can be summarised as follows.

- They can be built from local materials (Newcastle University and Environment Agency 2011).
- The feature is built across overland flow pathway to intercept flow (Newcastle University and Environment Agency 2011). It is designed to be an extension of the farming and land drainage scheme drainage regime (that is, the measures should not be viewed solely as flood engineering projects) (Quinn et al. 2013).
- They are often built in the lowest and therefore dampest part of a field.
- They do not have a significant impact on farming (Quinn et al. 2013).
- There are typically small in size (Quinn et al. 2013).
- The large volume of sediment removed from the flow can be returned to farmland (Newcastle University and Environment Agency 2011).
- They can be planted with native species in the appropriate locations. Plant species for swales, sediment traps and ponds are indicated in Duffy et al. (2016, p. 24).



Figure 4.36 Field treatment train

Source: Duffy et al. (2016)

Farm ponds

Farms ponds are usually created by excavation to form a small (temporary) ponding area that has a leaky barrier to ensure water is not contained but either spills over or is infiltrated into the soil. Rocks and vegetation are placed around the outlet to protect against erosion (Environment Agency 2012). The principles of designing permanent ponds are similar to those dry ponds and leaky barriers, and target fast overland flow. There is extensive literature on the design of ponds as SuDS and the evidence base for sizing, operating and maintaining ponds is set out in detail in the SuDS Manual (CIRIA 2015). There are many

inspiring examples to be seen on the Susdrain website.³¹ Figure 4.36 illustrates how CREW's practical design and build guide for RSuDS determines the size of a farm pond.

Wetland Design Equation

Total Wet Area of a Wetland = minimum 0.50% of field area

Pond Design Equation

Total Wet Area of a Pond = minimum 0.25% of field area

Worked Example 2:

A 13 ha arable field is adjacent to a watercourse in Fife. The field is in a rotation of root crops and winter wheat with frequent erosion of soil causing pollution of the watercourse. Therefore: Field area = 13 ha (equivalent to 130,000 m²) Percentage of field to be used for a Sediment Trap Bund = 0.25% (equivalent to 0.0025) Sediment Trap Bund plan area = 325 m² (130,000 x 0.0025) Sediment Trap Bund depth = 1.0 m Sediment Trap Bund volume = 325 m³ (325 m² x 1.0 m)

Figure 4.37 An example of the detailed design for ponds

Source: Duffy et al. (2016)

Permanent ponds

Any excavated soil (below-ground level) or a bund (above-ground level) can enable the formation of a farm pond. The only factor that makes it a permanent 'wet' pond is the constraint of the outfall or infiltration losses (perhaps by lining the pond). To obtain a permanent pond, more water must enter than leaves the pond – including during dry summer periods. Hence a local water balance is needed (Figure 4.38), which presupposes knowledge of inflows and outflows.



Z (depth) = $R + Q_{in} + Q_{ss}$ (subsurface) - E - Q_{out} - Infil (infiltration) - abs (abstraction)

Figure 4.38 Water balance for permanent ponds

Some permanent ponds need minimal excavation if the pond is deep enough or there is a shallow water table. Some permanent ponds can change in size and shape over a season and are designed to have multiple benefits (Natural England 2010b). As permanent ponds are conventionally used to trap pollution, they are not designed specifically for flood

³¹ <u>http://www.susdrain.org</u>

management. However, adding an extra leaky 'free board' component is a way of ensuring the flood storage capacity of the measures is increased and operates in the peak of events.

Key design guidelines include:

- irregular shape with islands and bars
- permanent pond: 1.2–1.8m deep, length to width ratio from 1.4:1 to 4:1
- inlet velocity 0.3–0.5m³s⁻¹ to prevent short circuiting
- overflow for extreme events is essential and must be designed to carry flows in excess of design water levels (from a 1% probability storm) to the downstream conveyance stream
- side slope 1:3 maximum
- volume of permanent pool = Vt (exceptionally 4Vt) where Vt = water quality volume of run-off from catchment
- freeboard above maximum level should be at least 0.3m

Dry retention ponds

Dry retention ponds should be designed to drain and empty completely within 5–20 hours to allow for further storage (Newcastle University and Environment Agency 2011). Basins are constructed by excavating a depression into the ground or by building an embankment at the bottom of a slop to impound stored run-off water (Environment Agency 2012).

Bunds are very important to dry retention ponds because the technique either includes infiltration or leaky barriers to increase the travel time. Bunds can be designed to overtop along their entire length, reducing the need for scour protection (Newcastle University and Environment Agency 2011). They can be either soil bunds, timber bunds or any type of leaky barrier. Bunds do not necessarily have to be a heavily controlled design (high bunds), but can be small-scale features designed and potentially installed by farmers or landowners themselves as 'broad and shallow' (<30cm) to deflect overland flow.

A RAF constructed in Belford (Figure 4.39) used locally sourced soil and boulders to form a bund over a known overland flow pathway that often erodes a gully in the field (Wilkinson et al. 2014). The bund also provides the landowner with a track to drive vehicles and machinery over the waterlogged zones of the field during wetter periods. It has a 0.22m diameter outlet pipe to allow it to drain in 8–10 hours to prevent seasonal waterlogging of productive fields.



Figure 4.39 Field bund storing water in a storm

Source: Wilkinson et al. (2014)

It has been observed in this project that features like this require a restricted height to avoid large localised scour and pressure increases in any field drains that may run beneath them (Wilkinson et al. 2014). Consequently, a height of 1m is set and any fast-flowing water should have rip-rap protection (Wilkinson et al. 2014). This type of feature is ideal for disconnecting fast flow pathways during the peak of storm run-off, with little impact on the farm, or in this case, have great benefit to the farmer.

Figure 4.40 shows the pilot pond full of water following a storm event, while Figure 4.41 illustrates its leaky nature.



Figure 4.40 Pilot pond at Belford full of water following a storm event in September 2008

Source: Wilkinson et al. (2010b, Figure 6a)



Figure 4.41 Leaky nature of the pilot pond at Belford

Source: Mark Wilkinson

Sediment traps

Sediment traps have a small temporary ponding area that is formed by excavation, and usually has rocks and vegetation around the gravel outlet to protect against erosion and overtopping (Environment Agency 2012, Duffy et al. 2016). The outlet pipe or overspill outlet should be designed to accommodate anticipated peak flows (Environment Agency 2012). The size of sediment traps depends on the soil type, intended run-off volumes to be intercepted and desired removal efficiency. In general, the larger the basin, the greater the

removal efficiency (Environment Agency 2012). Sediment traps should be located where a suitable area can be excavated or an embankment can be built across a swale, and where access can be provided for maintenance (Environment Agency 2012). According to CREW, sediment traps should be located downstream of a grass filter strip to help slow flows and reduce turbulence, so that sediments can quickly settle in the trap and enhance treatment effectiveness (Duffy et al. 2016).

Sediment traps are designed in a manner similar to that described in the SuDS Manual (CIRIA 2015) (Figure 4.42) by estimating flow rates and sizing the bunds appropriately. These designs are aimed at water quality management and therefore are not optimised for flood management. More often there is a more approximate method for the sizing and positioning of sediment traps, either in the corner of field or using a bund as a barrier to protect a channel or a road. Hence, the capacity of the bund may be suitable for average storms, but the bund may not work in extreme events and therefore may overtop (SEPA 2015). Overtopping means that flood flow and pollutants will enter the main channel or a road network, and cause problems downstream.



Figure 4.42 Example design outline for a sediment trap

Source: Duffy et al. (2016)

Swales

The Newcastle University and Environment Agency (2011) manual on RSuDS presents 3 designs of swales:

- Swale
- Enhanced dry swale: kept dry most of the time using a filter layer of soil over an underdrain
- Wet swale: soil is poorly drained and underdrains are not provided, so the swale acts as a linear wetland retaining water

Duffy et al. (2016) also describes a series of design options for swales that are appropriate for farms with pollution issues (Figure 4.43).

Swale: Typical Plan





Source: Duffy et al. (2016)

Swales are suitable for sites that are not flat or steeply sloping (Newcastle University and Environment Agency 2011). The groundwater must be more than 1m below the base of the swale if infiltration is required (Newcastle University and Environment Agency 2011). Swales are inappropriate for clean, coarse sandy soils as it is difficult to establish dense vegetation and prevent erosion even under very low flows (Newcastle University and Environment Agency 2011). CIRIA design concepts may also be useful to some aspect of swale design, especially if the swale is to be optimised for use in flood risk management (CIRIA 2015).

Swale base gradients should be shallow and no greater than 5° (1 in 20). They can also be built on steeper ground if designed to slow the flow and reduce the risk of erosion (Duffy et al. 2016). Check dams can enhance the performance of swales by maximising the retention time, decreasing flow velocities and promoting sedimentation (Newcastle University and Environment Agency 2011, Duffy et al. 2016).

The RSuDS Manual (Environment Agency 2012) mostly uses the key design factors from CIRIA (2004). It includes the following advice.

- Use Manning's equation to design slope, with an appropriate Manning's n value.
- Limit velocities to prevent erosion (typically 1–2m³s⁻¹ depending on the soil type).
- Maintain flow height below vegetation (typically 100mm).
- Minimum length of 30–60m, with a residence time greater than 10 minutes.
- Have a minimum base width of 0.6m and a maximum of 2.5–3m, unless a flow divider is provided to split the channel in two.
- Have a maximum side slope of 1 in 4. Check dams are recommended if the slope is greater than 3%.
- Infiltration, if required, should not be greater than 10⁻⁶m³s⁻¹.
- A mixture of vegetation should be planted.
- Trapezoidal channels are normal, but other configurations can also provide water quality improvements and may be easier to maintain.

Fiener and Auerswald (2003) completed a long-term landscape experiment to determine the potential of a grassed waterway to reduce run-off and sediment delivery from an agricultural watershed. The performance of the grassed waterway was found to depend on the length of the side slopes and the shape of its cross-section in the area of concentrated flow. An unmanaged grassed waterway with 2 times longer side slopes and a flat bottom had a higher run-off volume reduction (90%) compared with a cut and managed grassed waterway (10%); this was thought to be primarily caused by the differences in infiltration-induced sedimentation (Fiener and Auerswald 2003).

Maintenance and management

Run-off pathway management measures usually require low levels of maintenance. The most often cited measure is timely sediment removal for farm ponds, drainage pipes, sediment traps and swales in which landowners are encouraged to return trapped sediments to their fields (Newcastle University and Environment Agency 2011, Duffy et al. 2016). Sediment removal can depend on many factors including the characteristics of the catchment, soil health and the size of the run-off pathway management measure. Other maintenance such as vegetation and fence management, and regular inspection of measures for eroded or damaged areas may also be needed (Newcastle University and Environment Agency 2012, Duffy et al. 2016).

The study by Verstraeten and Posen (1999) found that, as a result of capturing sediment, retention ponds gradually fill and their water retention capacity is diminished. Consequently, the pond may not be able to store the run-off from an event for which it was constructed. This finding highlights the need for maintenance for these pond features to remain effective. If the removed sediment needs to be transferred to a dumping ground, further maintenance and costs would be required (Verstraeten and Pose, 1999).

Fiener et al. (2005) considered dry retention ponds that were constructed using on-farm machinery with almost no costs apart from those for the inlet raiser and transmitting pipes. Dredging was only necessary after the first year and completed using on-farm machinery at low costs. The small, earth-dammed retention ponds established at field borders were inspected regularly to identify any weaknesses. Reduction of peak run-off rates and low maintenance costs could only be achieved if regular siltation of the ponds was prevented due to effective soil conservation in the watershed (Fiener et al. 2005).

Lastly, it is vital that the design, construction, maintenance and management of a treatment train involve partnerships between regulatory bodies, farmers and landowners, local communities and other relevant stakeholders (Quinn et al. 2013). Experience at Belford has shown that a transparent and inclusive process avoids potential complications, encourages the uptake of the approach by all relevant stakeholders and can provide qualitative evidence for other local communities at risk (Quinn et al. 2013).

4.4.8 Research gaps

The literature review has highlighted a number of research gaps that need to be addressed to fully understand how run-off pathway management measures affect hydrological processes during floods and how their design and maintenance will have an impact on their effectiveness. The gaps listed below do not form an exhaustive list.

A critical gap remains in understanding the number of measures or storage (land area) needed at a catchment scale to slow the flow and reduce run-off to create an impact on flood risk downstream. A number of other gaps stem from this relating to the cost–benefit of these features in terms of lost productive land, and construction and maintenance costs, and in comparison with more traditional methods of flood management.

By using water level recorders, it can be established with confidence how much of the flow is being slowed and stored. However, there is a need to aggregate this local effect and to translate it to the larger scale.

Capturing data in ponds and RAFs during storm events is vital to enable evidence of RAF functioning to be presented to stakeholders; such data are also useful for suggesting improvements to their design and management. Data from the September 2008 flood level event at Belford was used to modify and optimise the function of a RAF (Nicholson et al. 2012). Thus, the ability to modify and optimise features and its impact provides useful evidence.

Research is also needed to address the specific flood flow design of swales, bunds and sediment traps to quantify how much flow rates are changed in large storm events. Equally when all the features are full, what is the aggregate impact of the features on flood flow? Good design guidance will lead to features being built that can target flood flow before it reaches the main watercourses.

There is need for a new breed of hydraulic models and aggregation schemes to reflect clusters of features and catchment models that can reflect the role of run-off pathway management. The conclusion from the study at Pontbren in mid Wales, which modelled a number of run-off pathway management measures, was that they can significantly reduce the magnitude of peak run-off at small catchment scale (Wheater et al. 2008). However, there is a gap in determining whether a network of permanent ponds can create a significant impact on flood flows in isolation from other run-off pathway management measures.

There is an opportunity for research to harmonise the costs and benefits of the run-off pathway management function for water quality, biodiversity management and flood management. Good metrics, such as cost per m³ for construction and saving by properties protected, are urgently needed to increase the confidence of stakeholders and provide measurable evidence.

Finally, a top-down analysis is needed that can determine for any catchment the amount of flood storage and the number and type of features needed to gain a specified peak flow reduction at a flood impacted site. This top-down analysis will allow a better investment plan to allow catchment NFM measures to be taken up. It may involve creating metrics to underpin soil and run-off (SPR and HOST) analysis at the catchment scale, so that practitioners can utilise their local knowledge more easily using existing models. Strategic planning for resource deployment and the issue of flow synchronisation will thus become more apparent.

Chapter 5. Coast and estuary management



5 Coast and estuary management

This chapter forms the literature review for coastal and estuarine WWNP interventions.

It begins with an explanation of the different interventions selected and summarises the most important R&D gaps uncovered by the review. Sections 5.2 to 5.6 then cover in detail each of the different WWNP interventions reviewed:

- 5.2 Saltmarshes and mudflats
- 5.3 Sand dunes
- 5.4 Managed realignment and regulated tidal exchange
- 5.5 Beach recharge/nourishment
- 5.6 Sediment bypassing

5.1 Introduction

5.1.1 What is WWNP in the context of coastal and estuarine management?

FCERM involves using a wide range of different types of intervention from constructing hard concrete defences though to work with nature to reduce flood risk, emergency planning and property level resilience. In the context of the coast and estuaries, Esteves (2014) suggests that for FCERM to be sustainable it should:

- avoid the need for increasing input of materials, labour and money to maintain them
- reduce risk to people and key assets
- · avoid preventable damage to or loss of natural habitats

This fits with the findings of the Pitt Review (Pitt 2008), which encouraged the government to work more with natural processes to reduce flood and coastal erosion risk rather than building ever taller concrete defences. FCERM interventions on the coast that work with rather than against coastal processes are more likely to achieve all 3 of these objectives in the long term than traditional forms of engineering.

To understand how best to implement WWNP requires a good understanding of coastal geomorphology and significant coastal processes (for example, waves, tides, longshore and cross-shore currents and storm surges). This literature review does not summarise these coastal processes as they are already covered in detail by others such as Pethick (1995), Bird (2008) and Masselink (2014). The purpose of this review is to establish 'what we know' and 'what we don't know' about the effectiveness of a range of different WWNP interventions in reducing flood risk.

On the coast and in estuaries, WWNP forms a spectrum of different types of intervention. These can range from doing nothing through to hard forms of engineering, as summarised below.

- **Doing nothing**. At its most minimalist, WWNP can mean doing nothing and letting nature restore itself. There are examples on the coast such as Brownsea Island in Poole Harbour in Dorset and Porlock in Devon where past embankments or walls are specifically not repaired so that they are breached and landward habitats restore themselves.
- **Prompted recovery**. This involves giving nature a nudge, helping it to restore a natural process that has been lost. An example of this on the coast would be removing groynes or other hard structures that are interfering with longshore drift, so as to allow natural processes to restore sediment supply downdrift.
- Working with Natural Processes. WWNP includes a suite of different measures, which are described in this chapter.
- **Hybrid approaches**. Hybrid schemes that include both WWNP and traditional forms of engineering are fairly common. One example is the Sandwich scheme in Kent, which involved repairing and raising a concrete tidal wall in the town alongside the creation of intertidal habitat, with the 2 elements of the scheme working together to reduce flood risk.
- Enhancing hard engineering. This approach involves enhancing traditional engineering measures mainly to provide ecological niches and habitats. On the coast it would include making indentations into coastal rock armour to create rock pool habitat. Although this is not common practice in the UK, it was piloted in the Shaldon scheme in Devon and has since been applied on part of the rock armour used in the Hartlepool scheme.
- **Traditional engineering**. This is not WWNP; in fact it often involves obstructing or working against natural processes to protect people and property from specific return period flood events. It can include many different interventions such as concrete walls, sheet-piled defences, rock armour and earth embankments. It also includes measures that manage natural processes such as offshore artificial reefs and groynes whose purpose is to modify the natural longshore and cross-shore movement of sediment.

WWNP on the coast can range from the restoration of saltmarsh, shingle beaches or sand dunes in a specific location to large-scale adaptation of coastal morphology to a form that requires minimum maintenance to sustain its FCERM functions (Nicholls et al. 2007). This can include the creation or management of morphological features such as ebb tidal deltas and spits, and optimising the configuration of coastal and estuarine morphology through managed realignment. It can also include 'traditional' coastal management measures such as beach recharge and beach bypassing, intended to hold the defence line on relatively heavily developed coasts. What all these approaches have in common is that they buffer the shoreline from incident wave and tidal energy. Some WWNP measures provide reservoirs of sediment, thereby generating a relatively self-sustaining solution that can accommodate dynamic changes. Such coastal features develop and evolve in response to changes in both forcing conditions and sediment supply. The standard of flood and coastal defence that they provide will therefore vary over time, but they also provide long-term resilience.

Various people have tried to define what WWNP on the coast means (see, for example, Bridges et al. 2013, 2015). Pontee et al (2016) described nature-based solutions as consisting either wholly, or partially, of natural features that are designed to offer or improve coastal protection such as:

- Fully natural solutions naturally occurring coral reefs, marshes and mangroves
- **Managed natural solutions** artificial coral/oyster reefs, renourished beaches and dunes, planted saltmarshes and mangroves
- **Hybrid solutions** combining structural engineering with natural features (for example, marsh–levee systems or dune–dyke systems)
- 'Environment-friendly' structural engineering vegetated engineering or bamboo sediment fences

As part of this review, discussions were held with the Environment Agency's Coastal Business Users Groups and Coastal Fisheries, Biodiversity and Geomorphology specialists to define which interventions this review should cover. To do this, a range of different types of coastal interventions were categorised according to whether they:

- resist coastal processes –includes hard engineering such as seawalls and revetments
- manage coastal processes (but not necessarily working with them) includes some groynes and beach profiling
- work with coastal processes –includes assisted recovery, creating/ restoring functioning coasts and the mimicking of natural processes to restore functioning coasts

Having categorised the different interventions (Table 5.1), the next step was to establish which helped to reduce FCERM. This process guided the choice of interventions to be covered in the literature review to ensure efforts were focused on those categorised as both WWNP and having an FCERM benefit. Those interventions selected for inclusion in the review were:

- enhancement/restoration/management of saltmarsh, mudflats and sand dunes (Sections 5.2 and 5.3)
- managed realignment of sea defences (including regulated tidal exchange) (Section 5.4)
- beach recharge/nourishment (including the 'sand engine') (Section 5.5)
- sediment bypassing (Section 5.6)

The science behind each of these selected WWNP interventions is described in the rest of this chapter.

5.2 Saltmarshes and mudflats

5.2.1 Understanding the science

Saltmarsh and mudflats have a valuable flood and coastal defence function along with their ecosystem and conservation importance. They also have roles in pollution control and the maintenance of water quality, fisheries, agriculture, recreation and tourism. These values are based on the interaction of their basic components (soil, water, flora and fauna), their physical shape (including channels and saltmarsh surfaces) and the assemblage of plants and animals they hold (Environment Agency et al. 2007).

Intervention type	Resisting natural	Managing natural	Working with natural	FCERM benefit?	Notes
	processes	processes	processes		
Enhancement of sea defence	Х				Can help extend the design life of existing hard defences, making them more resilient to the impacts of waves. Not a WWNP intervention.
Seawall	Х			Х	Resists natural processes of shoreline movement and erosion/accretion with hard reflective structures. Involves introducing artificial hard structures to the shoreline.
Rock revetment	Х			Х	Resists natural processes of shoreline movement and erosion/accretion with hard though less reflective structures. Involves introducing artificial hard structures to the shoreline.
Gabions	Х			Х	Resists natural processes of shoreline movement and erosion/accretion with hard though less reflective structures. Involves introducing artificial hard structures to the shoreline.
Groynes		Х			Encourages accretion of material of naturally transported material but artificially interferes with longshore drift. Manages natural processes of sediment accretion and movement, but is not WWNP because there are artificial interruptions to sediment movement.
Artificial offshore reefs (detached nearshore breakwaters)		X		Х	Encourages accretion of material of naturally transported material but artificially interferes with cross-shore movement of material. Manages natural processes of sediment accretion and movement, but is not WWNP because there are artificial interruptions to sediment movement.
Beach profiling		X		Х	Modifies the distribution of naturally occurring beach material in opposition to natural cross-shore processes. May be detrimental to shingle habitat. Manages natural processes of sediment accretion and movement. but is not WWNP because there are artificial interruptions to sediment movement.
Beach recycling		Х		Х	Transports naturally occurring material in opposite direction to longshore drift to address sediment shortage updrift. May be

Table 5.1 Categorisation of broad types of coastal FCERM interventions with respect to WWNP

Intervention type	Resisting natural processes	Managing natural processes	Working with natural processes	FCERM benefit?	Notes
					positive or detrimental to shingle habitat. Manages natural processes of sediment accretion and movement, but is to WWNP because there are artificial interruptions to sediment movement.
Beach recharge and sand engine			X	Х	Supplies new beach material on which natural processes will function. May be positive or detrimental to shingle habitat. Unsuitable material may negatively affect natural processes, habitat and FCERM function. Sand engine provides a large supply of beach material on which natural processes can then operate (experimental).
Beach bypassing		X	Х	Х	Transports material in same direction as longshore drift, for example, bypassing a groyne or estuary, to address sediment shortage downdrift. May be positive or detrimental to shingle habitat.
Saltmarsh and mudflat restoration		X	X	Х	Encourages natural accretion of mud (for example, by fencing, brushwood polders or planting) to create/enhance intertidal habitat that also has a FCERM function (may also involve artificial recharge).
Dune restoration		X	X	Х	Encourages natural accretion of sand (for example, by fencing, thatching or planting) and restoration of dune processes to create/enhance supratidal habitat that also has a FCERM function (may also involve artificial recharge).
Managed realignment		X	X	Х	Allows landward movement of shoreline in response to natural processes such as sea level rise. Creates intertidal habitat and more sustainable defence line, with intertidal habitat acting as part of a realigned defence.
Regulated tidal exchange		X	Х	(x)	Creates intertidal habitat but defence line still needs to be held. Can provide limited flood storage but direct FCERM function is limited to where used as a precursor to managed realignment.
Vegetated shingle			Х		Vegetated shingle is not considered to have a significant FCERM function. Unlike sand, shingle is not stabilised by vegetation as it

Intervention type	Resisting natural processes	Managing natural processes	Working with natural processes	FCERM benefit?	Notes
					cannot be moved by wind. Vegetated shingle will be eroded or moved by waves in the same way as unvegetated shingle. It is the mobile (largely unvegetated) zone of a shingle beach that plays a significant FCERM function by absorbing wave energy.
Estuarine reedbeds			X		Estuarine reedbeds can survive in brackish conditions and tolerate short periods of high salinity inundation. However, they are not able to withstand the regular wave action associated with open coasts. Usually they occur in upper estuaries that are sheltered from waves and subject to significant freshwater influence; or where protected from waves and full salinity by a shingle beach or other natural barrier. Consequently, estuarine reedbeds are not considered to represent a direct FCERM measure. Reedbeds may serve to trap or stabilise fine sediment, make sediment more cohesive and reduce currents by increasing surface roughness. Reedbeds could be an initial stage to build up sediment that will enable more characteristic and robust intertidal habitats to replace them as sea levels rise. These are benefits to the management of a 'biosedimentary' foreshore but no examples are known of where reedbeds have been created or enhanced specifically as a coastal FCERM measure. Consultation with the Environment Agency confirmed that reedbeds are considered too fragile to be effective in this role.
Coastal woodland			Х		Coastal woodland is found in a small number of locations, particularly in south-west England. Woodland is vulnerable to salinity and so cannot play an effective role in flood defence. Tree roots do stabilise topsoil against erosion. However, the rate of coastal erosion is generally a function of cliff recession at the toe where wave action is occurring, rather than being driven by topsoil erosion. Hence, coastal woodland is considered no more than peripheral to FCERM and is not included in this project.
Coastal heathland			X		Conservation measure with (generally) a peripheral significance to coastal defence. Coastal heathland occurs around the margins of Poole Harbour and the Isle of Purbeck in Dorset. Like woodland, heathland is vulnerable to salinity and so it cannot play an effective

Intervention type	Resisting natural processes	Managing natural processes	Working with natural processes	FCERM benefit?	Notes
					role in flood defence. Plant roots do stabilise topsoil against erosion. However, the rate of coastal erosion is generally a function of recession at the toe of the shoreline where wave action is occurring, rather than being driven by topsoil erosion. Hence, coastal heathland is considered no more than peripheral to FCERM and is not included in this project.
Natural reefs (for example, Sabellaria)			X		Conservation measure with (generally) a peripheral significance to coastal defence. Natural reefs (for example, Sabellaria) in British waters occur very low in the intertidal frame and are not considered to have any significant FCERM function, for example, in dissipating wave action. The Environment Agency is not aware of any examples of natural reef restoration or enhancement that have been carried out as an FCERM measure. Use has been made of oyster and mussel beds in the Netherlands to encourage accretion of soft sediment for flood and coastal risk management; this is covered under saltmarsh and mudflats (Section 5.2).
Eelgrass beds			X		Conservation measure with (generally) a peripheral significance to coastal defence. Eelgrass beds are similarly very low in the tidal frame (that is, the elevation range between the lowest and highest tides) and are not considered to have any significant FCERM function, for example, in dissipating wave action. They do stabilise soft sediments, and like reedbeds may make sediment more cohesive and reduce currents by increasing surface roughness. These are benefits to the management of a 'biosedimentary' foreshore but no examples of eelgrass restoration or enhancement that have been carried out as an FCERM measure were identified as part of this literature review.
Do nothing			Х		More sustainable defence line – no interference with natural processes but may be contrary to FCERM objectives.

In relation to FCERM, saltmarsh maintenance, restoration or enhancement is increasingly being considered as a means of making sea defences more sustainable and hence managing flood risk in estuaries. It also has the advantage of enhancing the conservation importance of a 'natural' and frequently designated priority and Biodiversity Action Plan habitat. Saltmarsh is a scarce and declining habitat in Britain, with 32,462ha in England, 5,800ha in Wales and 6,000ha in Scotland (JNCC 2016).

Saltmarshes receive more attention than mudflats as FCERM measures because they are higher in the tidal frame and have a vegetated surface, both of which contribute to attenuating waves. However, mudflats also play a role and the 2 habitats are interdependent, being usually found together. The interdependency of the saltmarsh and mudflat means that each one has a much better chance of survival if the other is present. Mudflats do exist without saltmarshes, but they are prone to erosion, especially on their upper levels and where waves are reflected from hard defences or natural cliffs. Saltmarshes can also exist without mudflats, but these are usually protected by some other form of defence (natural or artificial). If no protection is available at the seaward edge, then a marsh cliff will tend to form and may recede slowly landwards. The combined saltmarsh/mudflat landform is efficient at reducing wave action and should be considered as one unit by coastal managers (Environment Agency et al. 2007).

Saltmarshes and mudflats form part of estuary, coastal bay and often barrier beach systems. In order to develop and survive, they depend on processes and inputs from these larger systems. Both habitats are generally composed of mud or fine sand that settles out of suspension. For this to occur, the suspending water must move relatively slowly, with only low levels of turbulence. Once settled on the bed, the sediment can only accumulate and saltmarsh develop if the particles are not resuspended by wave or current action. The necessary low energy conditions for saltmarsh and mudflat development are normally encountered within the shelter of coastal bays, behind barrier islands or in estuaries. In each case, their location is primarily determined by the shelter afforded by the large-scale coastal morphology. However, within each of these environments, a wide range of energy levels can be found. The distribution and morphology of saltmarshes and mudflats is therefore determined not only by these coastal forms, but also by more local factors such as the tidal dynamics, sediment transport pathways, locally generated waves and the presence or absence of vegetation.

The 'Saltmarsh Management Manual' (Environment Agency et al. 2007) recognises 7 main types of saltmarsh:

- **Open-coast marshes** typically sandy systems with relatively exposed sandflats to seaward
- **Open-coast back-barrier marshes** sandy–muddy systems found on the sheltered, landward side of coastal barriers
- **Open-embayment marshes** marshes that fringe the edges of large tidal embayments with unobstructed entrances, and tend to be sandy
- Restricted-entrance embayment marshes typically mixed sandy–muddy systems, with the embayment partially closed off at the mouth by one or more spits or promontories
- Estuary-fringing marshes most commonly muddy and found in estuaries with little obstruction at the mouth
- Estuary back-barrier marshes found in estuaries behind barriers or spits at the mouth, often composed of mud overlying sand

• **Ria/loch head marshes** – marshes restricted to the drowned river valleys of the south-west, where pioneer to upper marsh occurs with transitions to woodland

Two prime functions of the networks of saltmarsh creeks are to:

- transport new sediment into the saltmarsh
- drain tidal water from the marsh surface on the ebb tide

Saltmarsh creeks serve to distribute tidal water and suspended sediment over the saltmarsh platform during the flood tide.

The restoration and enhancement of saltmarsh and mudflats seaward of existing defences can be carried out using a number of techniques. One approach used in relation to FCERM is to raise the intertidal level by recharging the intertidal area with sediment. Fine material is frequently available from dredging of estuaries and harbours for navigation purposes. The Environment Agency has produced a 'toolkit' for the beneficial use of dredgings in Essex and Suffolk (Environment Agency, undated a, undated b).

Another method for raising the level of intertidal mudflats is to encourage natural accretion of sediment to occur. This can be achieved using brushwood polders, rubble mounds or other barrier devices in the intertidal zone that slow down water currents and hence lead to the deposition of sediment.

Deltares (2013) reported that oyster reefs have been used in the Netherlands as a means of accreting sediment. Three large artificial oyster reefs ($200m \times 10m$) were built at 2 different places in the Eastern Scheldt. The boxes in these reefs had an area of $6m \times 2m$ and were 30cm high. Initial results showed that new oysters were quick to attach themselves to the gabions and the amount of silt behind the reef was increasing. Pilot studies on mussel beds showed that the bed affected the composition of sediments up to a distance of several hundreds of metres. Simulations performed in the laboratory also showed that the beds dissipated wave energy in shallow water.

Natural accretion on areas defended by seawalls can also be achieved by means of regulated tidal exchange.

5.2.2 Confidence in the science

There is abundant evidence from theoretical and practical studies (for example, et al. 1996, 2001, 2014, Möller and Spencer 2002) that saltmarsh plays a significant role in FCERM, particularly through its ability to:

- attenuate waves
- resist surface erosion
- accrete sediment until it is high in the tidal frame (usually close to mean high water spring level)

However, until recently there has been a lack of experimental measurements to quantify these effects.

In terms of practical design of FCERM schemes, a significant issue is the degree of confidence that can be placed in the sustainability of saltmarsh at a particular location. In other words, saltmarsh may be present seaward of a defence at the time the defence is designed, but engineers need to know whether it will still be present throughout the design life. There is a need for predictive tools to improve confidence in the presence of saltmarsh over timescales of a century, which is typically the design life of coastal defences.

5.2.3 Metrics of cost–benefit and multiple benefits

Saltmarshes provide a range of ecosystem services. The following ecological functions are considered to be of most significance (Environment Agency et al. 2007):

- flood and erosion control (through wave attenuation)
- maintaining water quality saltmarsh can act as a long-term (though not necessarily permanent) sink for a number of compounds considered to be pollutants including herbicides, pesticides, organochlorines, polychlorinated biphenyls and heavy metals; also nutrient cycling and sediment retention
- waste decomposition and disposal (microorganism processes and scavenging)
- habitat provision (spawning grounds for commercially exploited fish, habitats for plants and animals)

Van den Belt and Costanza (2011) summarised the services, processes and functions that are provided by estuarine mudflats and saltmarsh into 6 categories:

- food and raw materials
- coastal protection (that is, FCERM functions)
- habitat
- nutrient cycling and waste treatment
- climate regulation
- recreation, cultural and aesthetic services

The quantifiable benefits for each of these categories for saltmarsh creation schemes in the Humber Estuary are considered in Environment Agency (2007). As these are managed realignment schemes, the results are discussed in Section 5.4.

Eftec (2010) considered provisioning services, regulating services and cultural services, quoting an economic valuation for saltmarsh –, including habitat, recreation and nutrient storage but excluding carbon storage and flood risk mitigation – in the range of £200 to \pounds 4,500 per hectare per year. This is possibly too wide a range to be useful and suggests that more site-specific studies are needed.

Saltmarshes sequester carbon and by doing so pay a role in regulating climate change. Beaumont et al. (2013) estimated that:

- saltmarsh vegetation and soils across the UK sequester 6,000 tonnes of carbon per year
- the economic value of carbon sequestration by saltmarsh is in the range of £35 to £118 per hectare per year

The sequestration capacity of current coastal habitats over the period 2000 to 2060 is estimated to be in the region of £1 billion (3.5% discount rate). However, if current trends of habitat loss continue, the capacity of the coastal habitats to both sequester and store carbon dioxide will be significantly reduced, with a reduction in value of around £0.25 billion (3.5% discount rate).

Saline wetlands represent one of the largest components of carbon sinks globally, storing in excess of 45 million tonnes of carbon annually (Chmura et al. 2003). Therefore, the storage of carbon by saltmarsh in managed realignment sites could potentially be an important sink for atmospheric carbon dioxide. It is also reported by Mitsch and Gosselink (2007) that since

saltmarshes are one of the most productive ecosystems in the world (storing up to 3.9kg of carbon per cm² per year) and one that is widely distributed, saltmarshes sequester millions of tonnes of carbon per year.

Holm et al. (2016) quoted annual soil carbon sequestration rates of natural marshes of ~200g of carbon per m² per year (7.3tCO₂e per hectare per year). However, methane emissions from wetlands offset more than half the benefit, as they are equivalent to an estimated $4.1tCO_2e$ per hectare per year.

Given the anoxic nature of marsh soils, it has been suggested that carbon sequestered by saltmarsh plants is often lost from the short-term carbon cycle (10–100 years) to the long-term carbon cycle (1,000 years) as buried, slowly decaying root material (Mitsch and Gosselink 2007, Mayor and Hicks 2009). However, this critical ecosystem service generated by saltmarshes is beginning to be recognised by managers, economists and scientists (van den Belt and Costanza 2011).

A study by Shepherd et al. (2007) investigated the effects and economics of managed realignment sites on the cycling and storage of nutrients, carbon and sediments in the Blackwater Estuary in Essex. The cost–benefit analysis results showed that over the 50–100 year time frame, the value of habitat creation (including nutrient storage) and carbon sequestration was sufficient to make large-scale managed realignment cost-effective. Evidence for the Alkborough managed realignment site suggests that the site provides an annual benefit of £14,553 from carbon sequestration (Environment Agency 2009).

5.2.4 Effectiveness/performance

Significant factors that control the degree of wave dissipation over saltmarshes include water depth, vegetation canopy height, vegetation canopy density, wave height and wave period. The effectiveness of saltmarsh in FCERM is a focus of current research. Estimates can be derived from 3 main approaches:

- theoretical, for example, using numerical modelling
- practical measurements of wave attenuation made in controlled conditions (that is, wave tanks)
- practical measurements of wave attenuation made in field conditions (that is, estuaries or open coast)

A number of older literature sources are frequently quoted giving estimates (mostly based on theoretical considerations) for the effectiveness of saltmarsh as a contribution to coastal flood risk management, as in the following 2 examples.

- Saltmarsh vegetation can diminish wave heights by up to 70% and wave energy by over 90% (Bird et al. 2000).
- Approximately 80m width of saltmarsh in front of a flood defence structure can save about £4,600 per metre in additional wall protection (Empson et al. 1997).

The reliability of these older estimates is unclear as they are based on numerical modelling and lack supporting field observations and calibration (Möller et al. 2001). They assume that wave energy reduction is caused by shoaling/breaking processes and by frictional losses, in accordance with physical and numerical model results and general wave theory. These approaches have failed to address all the effects of vegetation density, rigidity and roughness, saltmarsh topography, substrate and inundation depth on wave energy dissipation. More recent work has been carried out by the University of Cambridge's Coastal Research Unit (CCRU), including field and laboratory measurements of wave attenuation over saltmarsh surfaces. The following 3 studies have provided useful information.

An experimental assessment of wave dissipation under storm surge conditions used a 300m long wave flume tank containing a transplanted 40m section of natural saltmarsh. This found that the presence of marsh vegetation caused considerable wave attenuation, even when water levels and waves were highest (2m water depth and waves up to 0.9m). From a comparison with experiments without vegetation, it was estimated that up to 60% of observed wave reduction can be attributed to vegetation (Möller et al. 2014). It was also found that, although waves progressively flatten and break vegetation stems and thereby reduce dissipation, the marsh substrate remained stable and resistant to surface erosion under all conditions. The effectiveness of storm wave dissipation and the resilience of tidal marshes even at extreme conditions suggest that saltmarsh ecosystems can be a valuable component of coastal protection schemes.

An array of 3 bottom-mounted pressure transducers (placed ~200m apart along a shorenormal transect centred on the sandflat/saltmarsh transition) was used to measure changes in wave characteristics across sandflat and saltmarsh on the Norfolk coast. Pressure readings were taken at a frequency of 5Hz over periods of 5 and 7 minutes at different times during the tidal cycle over a range of tides between September 1994 and May 1995 (Möller et al. 2001). Analysis showed a consistent energy decrease of between 47.4% and effectively 100% across the saltmarsh section of the transect. This differed significantly from the much lower wave energy reduction (1.9–55.3%) across the sandflat section of the transect. The reduction in wave energy and significant wave heights was only weakly related to water depth across the sandflat, but more strongly related to water depth across the saltmarsh. The results suggest that saltmarshes are extremely effective in buffering wave energy over the range of water depths and incident wave energies investigated. The increased surface roughness of saltmarshes is likely to be most effective in reducing wave energy at low to intermediate water depths (<1.1m) or during conditions of high incident waves.

A 10-month long wave/tide dataset from 2 sites on the Dengie Marshes in Essex was used by Möller and Spencer (2002) to look at the effect on wave height and energy dissipation of:

- marsh edge topography
- marsh width
- inundation depths
- seasonal changes in marsh surface vegetation cover

Directional waves and water levels were recorded at 21 locations across both shallowsloping and cliffed intertidal profiles. In addition, changes in marsh surface vegetation cover and composition were recorded on a seasonal basis. Wave height attenuation over 310m of the shallow-sloping profile averaged 92% over the monitoring period. The most rapid reduction in wave heights occurred over the most seaward 10m of permanent saltmarsh vegetation, where wave height attenuation averaged 2.1% and 1.1% per metre at the shallow-sloping and cliffed sites respectively. Across the mudflat and the saltmarsh as a whole, wave height dissipation rates were significantly lower with an average of 0.1% and 0.5% per metre respectively. The presence of a saltmarsh cliff increased average wave heights by up to 0.5% per metre. Observed wave height attenuation showed a seasonal pattern at both sites (average wave energy attenuation near the marsh edge was highest in September to November and lowest in March to July) and appeared to be linked to the cycle of seasonal vegetation growth.

Results from these studies are being used to refine parameters in numerical models, which in theory enable the design of sea defences to take into account the wave attenuating properties of saltmarsh. However, an obstacle to achieving savings in sea defence construction arises from uncertainty as to the sustainability of saltmarsh over the design life of the defence, which is typically 100 years. Saltmarsh may be lost due to erosion or sea level rise, and these factors are also the subject of investigation by CCRU:

Wave flume experiments show that the vegetated marsh surface is resistant to erosion. Loss due to erosion therefore arises from undercutting and retreat of the edge of the marsh. Research is being conducted to develop predictive tools based on satellite data, Environment Agency aerial photography and LiDAR imagery to predict likely marsh stability. This is then being used, together with contextual environmental information, to predict marsh edge type and stability with a purpose-designed state of the art statistical (Bayesian) model (T. Spencer 2016, personal communication). Outputs from this study are expected to be available in early 2018.

The effect of sea level rise depends on the availability of sediment. Experience suggests that, where sediment supply is sufficient, in general saltmarshes will accrete to keep pace with sea level rise.

Experiments by Ford et al. (2016) on the erosion of saltmarsh soils in a flume found that soil stabilisation and root biomass were positively associated with plant diversity.

CCRU is involved in the development of 2 projects to create predictive tools for FCERM managers and designers. The first is Foreshore Assessment using Space Technology (FAST), an EU-funded 4-year project³² that is developing a web tool (code-named MI-SAFE) that will allow the user to find out the degree to which a particular foreshore is likely to act as a natural coastal protection, both in terms of wave dissipation and surface stability. The tool makes use of the revolutionary new Sentinel suite of environmental satellites that allow, for the first time, coastal surface attributes to be derived from space at a scale of around 10m x 10m. CCRU is, together with EU partners in the Netherlands, Romania and Spain, providing all the scientific evidence while Deltares in the Netherlands is building the web tool. The second, Risc-Kit, is also an EU-funded project.³³ It has developed a toolkit approach for improving flood risk preparedness and prediction at complex case study sites, such as North Norfolk, where the natural landscape contributes to flood and erosion risk reduction. CCRU has been involved in the numerical modelling of surge/waves for that case study site, proving the contribution made by the natural features, as well as acting as the linking partner to allow the project to benefit from contact with a range of stakeholders. Together with the Environment Agency, consideration is being given as to how the new Risc-Kit tools might be incorporated into existing flood forecasting and assessment methods and which of these new tools might complement existing Environment Agency provision.

Vuik et al. (2016) analysed the effect of vegetation on wave damping under severe storm conditions, based on a combination of field measurements and numerical modelling. The field measurements of wave attenuation by vegetation were performed on 2 saltmarshes with 2 representative but contrasting coastal wetland vegetation types, cordgrass (*Spartina anglica*) and grassweed (*Scirpus maritimus*). The former is found in salty environments, whereas the latter is found in brackish environments. The wave measurements described in this study added to the range of observations with the highest water depths (up to 2.5m) and wave heights (up to 0.7m) presented in the literature so far. A numerical wave model (SWAN) has been calibrated and validated using the new field data. It appeared that the model was capable of reproducing the observed decay in wave height over the saltmarsh. The model has been applied to compute the reduction of the incident wave height on sea defences for various realistic foreshore configurations and hydraulic loading conditions.

The efficiency of vegetated foreshores in reducing wave loads on the defence was also investigated by Vuik et al. (2016). Wave loads were quantified using a computed wave run-

³² http://www.fast-space-project.eu/

³³ http://www.risckit.eu/
up height and wave overtopping discharge. The outcomes show that vegetated foreshores reduce wave loads on coastal defences significantly, and also for the large inundation depths that occur during storms and with the vegetation being in winter state. The effect of the foreshore on the wave loads varies with the wave height to water depth ratio on the foreshore. For example, if the depth on the foreshore is limited to just 1.0m, the wave run-up is reduced by 60–100%, and the wave overtopping discharge diminishes to negligible amounts. For larger water depths, the influence of vegetation becomes more distinct. A typical design water level is in the order of 5m above mean sea level, which is 3m above the saltmarsh surface. Where wave run-up under these conditions is only reduced by approximately 20% (0.6m) for a 400 m wide, bare foreshore, the same foreshore covered by vegetation resembling *Spartina anglica* in its winter state reduces the wave run-up by 55% (1.8m). Wave overtopping discharges still have significant values for bare foreshores in case of large water depths, whereas the presence of vegetation prevents the occurrence of overtopping.

Vuik et al. (2016) concluded that the presence of vegetation on the foreshore extends the range of water depths for which a foreshore can be applied for effective reduction of wave loads, and prevents intense wave breaking on the foreshore from occurring. This research demonstrates that vegetated foreshores can be considered as a promising supplement to conventional engineering methods for seawall reinforcement.

Narayan et al. (2016) reviewed the effectiveness, costs and coastal protection benefits of natural and nature-based defences. They analysed data from 69 field measurements in coastal habitats globally and examined measures of effectiveness of mangroves, saltmarshes, coral reefs and seagrass/kelp beds for wave height reduction (Figure 5.1).



Figure 5.1 Wave reduction by 4 different habitat types

Notes: Absolute wave reduction extents plotted against incident wave height for range of habitats. Excludes measurements that do not report incoming wave heights. Source: Narayan et al. (2016)

Examining the costs and coastal protection benefits of 52 nature-based defence projects, they estimated the benefits of each restoration project by combining information on restoration costs with data from nearby field measurements. The analyses of field measurements showed that coastal habitats had a significant potential for reducing wave heights which varied by habitat and site. In general, coral reefs and saltmarshes had the highest overall potential. Habitat effectiveness was found to be influenced by:

- ratios of wave height to water depth and habitat width to wavelength in coral reefs
- ratio of vegetation height to water depth in saltmarshes

The comparison of costs of nature-based defence projects and engineering structures showed that saltmarshes and mangroves can be 2–5 times cheaper than a submerged breakwater for wave heights up to 0.5m and, within their limits, become more cost-effective at greater depths.

5.2.5 Key case studies

Table 5.2 lists 17 sediment recharge schemes undertaken in Great Britain. Data are sourced from the Online Marine Registry (OMReg).³⁴ Of these 17 schemes:

- 15 include 'habitat creation' or 'habitat enhancement' or 'habitat restoration' among their objectives
- 2 include 'compensation' or 'compensatory habitat' among their objectives
- 2 include 'development need' among their objectives
- 2 include 'mitigation' among their objectives
- 1 includes 'demonstration' or 'pilot' among its objectives
- 1 gives 'unknown' as its objective

5.2.6 Funding

Saltmarsh and mudflat management/restoration can potentially be funded through Flood Grant-in-Aid where FCERM is a primary objective.

Where the recharge of mudflats is primarily driven by the need to find a beneficial use for dredgings, funding will normally be provided by the dredging operation.

Where saltmarshes are enhanced or restored for biodiversity gain, funding may be available from the Higher Tier of Defra's Countryside Stewardship. This can pay feasibility, capital and management costs in relation to eligible habitat creation, including:

- CT6: Coastal vegetation management supplement
- CT5: Creation of intertidal and saline habitat by non-intervention
- CT3: Management of coastal saltmarsh

³⁴ <u>http://www.omreg.net/</u>

Scheme name	Location	Total area	Year opened	Habitat created	Technique	Primary purpose of recharge
Allfleet's Marsh	Wallasea Island, Essex	23ha	2006	Saltmarsh	Level of managed realignment area raised using maintenance dredgings	Compensation, habitat creation
Bedlam's Bottom	Medway, Kent	Not known	1996	Not known	Trickle charge feeding of lower intertidal with fine muddy dredged material	Habitat enhancement, demonstration/pilot
Boiler Marsh	Lymington, Hampshire	1ha	Not known	Not known	Maintenance dredged silt placed on an eroding area of saltmarsh	Mitigation, habitat enhancement
Cindery Island	Colne Estuary, Essex	Not known	Not known	Not known	Silts dredged from the harbour placed in disused oyster pits to recreate saltmarsh	Habitat enhancement
Fambridge (Westwick Marina)	Crouch Estuary, Essex	Not known	2001	Not known	Dredged fine muddy sediment pumped onto the adjacent marshes	Habitat enhancement
Foulton Hall and Stone Point	Hamford Water	Not known	1998	Not known	Dredged gravel and sand deposited at low water as part of saltmarsh creation and restoration	Not known
Levington	Orwell Estuary, Suffolk	Not known	Not known	Not known	20,000m ³ dredged material is pumped annually onto foreshore and retained using wattle hurdles or faggots to raise intertidal level	Economic need/development, habitat enhancement
Lymington Yacht Haven Marina	Lymington Estuary, Hampshire	0.75ha	2012 to 2013	Not known	2,000m ³ of maintenance dredged silt material was placed to counteract ongoing marsh erosion and as mitigation for habitat damage	Mitigation, habitat enhancement

Table 5.2Sediment recharge schemes in Great Britain

Scheme name	Location	Total area	Year opened	Habitat created	Technique	Primary purpose of recharge
Maldon (Hythe Quay)	Blackwater Estuary, Essex	Not known	Ongoing since 1993	Not known	2,000m ³ of maintenance dredged material placed annually to nourish saltmarsh	Economic need/development, Habitat enhancement
North Shotley	Orwell Estuary, Suffolk	Not known	Not known	Not known	15,000m ³ of dredged gravel and silt was placed onsite, retained using clay and gravel bunds	Habitat enhancement
Parkeston Marshes	Stour Estuary, Suffolk	Not known	Not known	Not known	250,000m ³ of dredged sand was deposited to arrest erosion of the foreshore and restore wetland	Habitat restoration
Pewet Island	Blackwater Estuary, Essex	Not known	Ongoing since 1992	Not known	Sand/shingle rainbowed to raise foreshore level	Habitat restoration
Parkstone Yacht Club	Poole Harbour, Dorset	0.65ha	Not known	Not known	Existing intertidal sediments and dredged silts and sands were placed within a rubble mound breakwater and sheet piling	Habitat creation
South Shotley	Orwell Estuary, Suffolk	3ha	1998	Not known	22,000m ³ of maintenance dredged sediment (mostly silt) was placed and retained using 75,000m ³ retaining bund of coarse gravel	Habitat enhancement
Titchmarsh Marina	Hamford Water, Essex	Not known	1998, 1999, 2002, 2007	Not known	Dredged material placed on intertidal area to raise level for saltmarsh development	Habitat enhancement
Trimley realignment	Orwell Estuary, Suffolk	16.5ha	1998 and 2001	Not known	57,000m ³ of dredged material pumped to raise level of managed realignment site	Compensation, habitat creation

5.2.7 Design, management and maintenance

Techniques for managing saltmarsh can be divided into 3 main groups.

- Improve ecological characteristics or to restore the marsh:
 - grazing management
 - vegetation planting
 - pollution source control and management of pollution events
 - management of freshwater input/drainage
 - management of access

These approaches relate primarily to nature conservation interests and are therefore outside the scope of this report.

- Managing erosion/accretion:
 - sedimentation fields
 - intertidal recharge
 - vegetation planting
 - sediment source control
 - other hard engineering techniques such as breakwaters or groynes
 - other erosion management techniques such as brushwood and drainage furrows

These approaches are of primary interest to FCERM and are covered below.

- Creating new saltmarsh to landward:
 - managed realignment
 - regulated tidal exchange systems

Hofstede (2003) described a number of saltmarsh management techniques found to be beneficial to FCERM on the Schleswig-Holstein coast of the Wadden Sea, Germany. These are outlined below.

Drainage furrows within mudflats just seaward of saltmarshes improved the drainage, consolidation and accretion of the upper layer of mudflats and therefore the ability of pioneer vegetation such as *Salicornia* to colonise. This was found to result in a seaward shift of the outer edge of saltmarsh by up to 200m. In order to function, the furrows need to be cleaned on a regular basis. The main drainage furrows can be as wide as 3.2m and 0.4m deep; the smallest ditches are about 0.3m wide and 0.15m deep.

Groynes constructed in high mudflats are another traditional technique to enhance saltmarsh accretion and stabilise existing marshes. They work by reducing wave, tidal current and storm surge velocities to create an artificial low energy environment, encouraging sedimentation and hindering erosion. Groynes are usually constructed of brushwood wedged between wooden poles. In one instance, 40cm of sedimentation was recorded in a single year after construction of such a groyne field.

Clay dams run from the seawalls to the outer edge of the saltmarsh, spaced about 400m apart. The primary purpose of these dams is transport routes for maintenance and disaster

recovery vehicles, but they also reduce wave action and longshore currents. These structures are not considered to constitute WWNP.

Saltmarsh turfs (consisting of halophytic vegetation) are used to strengthen the lower outer surface of seawalls against erosion.

Grazing is believed to increase the shear strength of subsoil and to diminish the amount of floating organic matter washed up after winter storms. However, research has found that even without grazing, saltmarsh shear strength is still sufficient to resist erosion.

Regional saltmarsh management plans have been put into effect by the regional government of Schleswig-Holstein, using a combination of groyne fields and drainage furrows to extend saltmarshes seaward and to create new areas of saltmarsh. An initial groyne field is typically 200m deep from the shore and, if successful, a second phase may be constructed to the seaward of this, sometimes followed by a third phase. A 'developing' saltmarsh is considered to become an 'existing' saltmarsh when it reaches a width of 200m.

5.2.8 Other requirements

Natural England has proposed that estuarine saltmarsh creation should take account of Estuary Regime Theory (Natural England 2016). This is based on the principle that tidal energy controls channel size by promoting erosion to expand the channel if the estuary is too narrow, or accretion to make the channel smaller if it is too large. Equilibrium is an ideal state where there is a balance between erosion and accretion, so despite adjustments over time, the overall form is stable. Studies of many estuaries around the world have demonstrated a relationship between cross-sectional area at the mouth and the tidal prism (that is, the volume of water in an estuary between mean high tide and mean low tide). Habitat creation should take account of the observed estuary form and compare it with the predicted form at different points from the mouth to the upper limits. To be sustainable, saltmarsh creation should aim to move estuaries closer to equilibrium.

5.2.9 Research gaps

There is theoretical, modelling and experimental evidence that saltmarsh, and to a lesser extent mudflat, can play a significant role in attenuating waves, thereby reducing the erosion and overtopping of sea defences. This can potentially lead to substantial cost savings on defence construction and maintenance. However, there are limited data from field measurements in the UK on the realisation of savings in flood defence construction. Field measurements of wave attenuation are difficult to make, while a practical impediment to realising savings is uncertainty over the longer term sustainability of saltmarsh at a particular location. Further evidence and guidance are needed to enable the benefits of saltmarsh to be incorporated into sea defence design.

Most projects to create or restore saltmarsh seaward of existing defences in the UK have involved small-scale fence or brushwood polder construction to encourage accretion or have been driven by the need to find beneficial use for dredged material by placing in the intertidal zone. Most have been primarily oriented to habitat and biodiversity gain. There is little evidence of any large-scale, strategic approach to saltmarsh creation/restoration seaward of existing defences primarily for FCERM. Evidence from the North Sea coasts of Germany and the Netherlands suggests that such approaches can be successful and would repay further investigation for applicability in UK. However, it was not possible within the resources of this project to obtain the information for a case study on this.

Important research gaps include the following.

- There is a need for more measurements of wave attenuation by vegetated surfaces (saltmarsh) in the field as opposed to experimental wave tanks.
- Greater confidence is needed in the sustainability of saltmarsh over the life of FCERM schemes and approaches to adapting FCERM design.
- Techniques and data availability for ecosystem services valuation need to be improved.
- Practical techniques are needed for estimating the degree to which a particular foreshore is likely to act as a natural coastal protection. This is being addressed by the FAST project.
- Improved techniques are needed for flood risk prediction and preparedness at sites where vegetated foreshores contribute to flood and erosion risk reduction. This is being addressed by the Risc-Kit project.
- Evidence from large-scale saltmarsh/mudflat restoration projects in Germany and the Netherlands is needed to develop comparable pilot projects in UK.

5.3 Sand dunes

5.3.1 Understanding the science

Sand dunes form where there is a supply of sand over a wide foreshore that has sufficient time to dry between tidal inundation, a backshore area of low relief, and predominantly onshore winds for at least part of the year.

Dunes can be extremely important because of their flood defence properties, both by providing a barrier to inundation and by releasing sand during storm conditions to reduce wave action. Where appropriate conditions exist for a healthy functioning dune system, this natural coastal defence can be highly cost-effective compared with the expense incurred in maintaining alternative seawalls and other hard coastal defences (Skelcher 2008). However, few data are available to demonstrate the economics of dunes compared with other defences.

Initially wind-blown sand accumulates around small objects such as clumps of seaweed, driftwood or other debris cast along the strandline. Beach cleaning by local authorities can be detrimental to this process by removing the nuclei for dune generation (Dornbusch, personal communication). Ideal conditions for the transport of sand from a beach to the dunes occur after waves have deposited sand on the upper beach and intertidal foreshore. At low tide, the sand dries and onshore winds can carry substantial volumes of sand onto the dunes. Dunes can also sometimes form under offshore winds (Lynch et al. 2009). Once formed, low hills of loose sand are then colonised by salt-tolerant, pioneer plants that both increase the resistance of the surface layer of sand to wind erosion and reduce the wind speeds. The embryo dunes, or foredunes, will continue to grow unless they are destroyed by wave action at high tide levels. The main pioneering colonisers of loose sand include sand couchgrass (*Elytrigia juncea* and lyme grass (*Leymus arenarius*). These species are able to withstand short periods of immersion by seawater and have long roots, rhizomes and runners that are able to bind the surface grains and extend the vegetation cover laterally (Pye et al. 2007).

As the foredunes grow vertically above the level reached by waves, they are colonised by marram (*Ammophila arenaria*), which thrives on continual burial by the blown sand deposits. The marram-dominated mobile dunes remain unstable due to the exposure of sand between the clumps of vegetation. If new foredunes develop in front of the mobile dunes, the marram

ceases to be supplied with fresh sand deposits, and other species colonise and stabilise the dune surface. The composition of the formed fixed dunes then varies depending on the soil chemistry but is often quite calcareous. Fixed dunes are usually dominated by red fescue (*Festuca rubra*) and support a much higher diversity of species than mobile dunes. As the fixed dunes age, they lose lime and develop a flora with common bent (*Agrostis capillaris*) and patches of heather (*Calluna vulgaris*), which is known as dune heath (Pye et al. 2007).

Dune slacks are damp, low-lying areas between dune ridges which can be particularly rich in plant species including creeping willow (*Salix repens*), sedges, rushes, orchids and mosses.

Sand dunes are a scarce habitat in Britain; JNCC gives areas of 11,897ha in England, 8,145ha in Wales and 50,000ha in Scotland (JNCC 2016). Most sites are less than 100ha in size and only a few exceed 500ha (Pye et al. 2007). Sand dunes provide an important habitat to a wide range of wildlife, including a large number of species that are incapable of surviving in any other habitat – many are of national or international significance.

Sand dunes have considerable flood risk management significance, for example, on the Sefton Coast (Lancashire), Lincolnshire and north-east Norfolk where extensive low-lying areas are protected from coastal flooding by naturally occurring dune ridges. Their importance in this regard lies primarily in their function as barriers to coastal flooding. Dune systems are especially important where they protect high density residential or industrial developments, high-grade agricultural land or habitats of international conservation importance. Compared with many other forms of defence, dunes are less visually intrusive, have greater value for wildlife and recreation, and are able to respond more readily to changes in environmental forcing factors such as climate and sea level change, sediment supply conditions (Pye et al. 2007).

The importance of sand dunes in flood protection depends on their location and the height and width of the dunes relative to the assets located inland (SEPA 2015). Dunes can act as a natural dynamic coastal defence, absorbing wave energy and releasing sediment to the beach during storms, and rebuilding by wind action during periods of fair weather. Continuous dune ridges provide the best flood protection value, with blow outs and/or low or narrow points creating areas more vulnerable to flooding. Dune systems less than 5m wide and/or less than 2m high can be considered to have little flood protection value on an open coast, as it is easily possible for such dunes to be eroded or severely overtopped in a single storm. Vegetated dunes are more stable, effectively trapping sand and binding it together with root systems, creating a more effective flood protection structure.

Virtually all dune fields in England and Wales have formed entirely in the last 5,000–6,000 years and, in most places, the present dune topography is less than a few hundred years old. Many sites still had extensive areas of bare sand as recently as the 1970s, largely as a result of human activities. Dune stabilisation measures since the 1950s, and particularly in the 1980s and 1990s, have stabilised most dune fields to a high degree. Areas of Aeolian activity are now restricted mainly to sections of eroding coast and a few inland blowouts, which have remained active due to local wind acceleration and increased turbulence (Pye et al. 2007).

Most dune systems in England and Wales are composed of quartz sands. Marine carbonate is important only in some systems in Devon, Cornwall and south-west Wales. The main sources of sand in the past were marine reworking of glacial sediments on the seabed and in coastal cliffs. These sources are much less significant at the present time. Increased storminess and rising sea level are likely to cause more widespread erosion, leading to redistribution of existing coastal sediments. Accretion can be expected at the downdrift ends of sediment transport cells, but dunes at the updrift ends will experience accelerated erosion and greater risk of breaching/overtopping (Pye et al. 2007).

Approximately 35% of the total dune frontage in England and Wales has experienced net erosion or is protected by hard defences, 35% has experienced net stability and 30% net seawards accretion (Pye et al. 2007). The extent of frontal dune erosion may increase in the next century as a result of increased storminess and sea level rise. This may have negative impacts on the extent of some dune habitats and the effectiveness of dune systems as flood defences. However, the consequences of such changes will vary from location to location, reflecting differences in natural processes and beach–dune sediment budgets.

Sand dunes are inherently dynamic, undergoing periods of erosion and accretion. Changes occur in response to weather, climate, ecological succession and human management. Across the UK, there has been a trend in recent decades towards increased stabilisation of dune habitats and a consequent reduction of mobile sand with its associated pioneer habitats. Grassy sward development and scrub invasion now pose a serious threat to the existence of the habitats and species for which sites were designated as being of national and international conservation importance (Pye et al. 2014). Pye and Blott (2012) estimated that, in Welsh sand dunes, 79% of the bare sand present in the 1940s/1950s had become vegetated by 2009. Although changing climate appears to have contributed significantly to this trend, the situation has been made markedly worse by human activities, including forestry development, dune management practices and the implementation of restrictive conservation designations.

In the past, management of sand dunes for FCERM has often taken the form of artificial stabilisation (for example, planting pines). In some locations, seawalls have been constructed at the toe of the dunes, which severs the connection between the mobile and pioneer dunes, as well as reduces the supply of sand. More recent approaches focus on maintaining or restoring the supply of sand to encourage dunes to be sustained through natural processes. Excessive stabilisation of dunes, whether through artificial means or as a result of ecological succession, can be detrimental to long-term FCERM as well as nature conservation, since it reduces the ability of the dunes to evolve under forcing factors (such as sea level rise and changes in nearshore bathymetry) and makes them less resilient to coastal change. Pye and Blott (2012) recommended that the area of pioneer/mobile dune habitat and bare sand should be 30–40% in a healthy, dynamic dune system. However, maintaining healthy mobile dunes requires space for them to migrate, and where none is available (for example, because there is an urban area in the immediate dune hinterland), fixing the dunes in position may be the only available FCERM option.

In recent years, most dune restoration measures have been carried out for nature conservation reasons rather than as FCERM measures. However, the restoration of natural processes to dune systems can have FCERM benefits.

Between 2003 and 2009, the National Trust restored South Milton Sands in Devon, a heavily used 2ha sand dune site with a small beach and extensive car parking (Ford et al. 2009). The wooden piling defences constructed in 1990 were at the end of their lifespan and thought unsustainable considering the erosion at the site. The designed scheme removed the failing defences and reprofiled the dunes, which allowed the dunes to erode and build according to natural processes. Once groundwork was completed, local people helped plant the marram grass (*Ammophilia arenaria*) on the dunes. Following consultation, the National Trust agreed to maintain a small area of defence to an existing slipway for a 10-year period and a small amount of rock armour was provided to protect adjacent apartments.

At Fylde Sand Dunes in Lancashire, detailed recommendations have been made by the Fylde Sand Dune Project Steering Group for dune management in the interests of nature conservation, flood defence and recreation (Skelcher 2008). These include measures to encourage mobile sand accretion using brush wood or old Christmas trees to trap sand. The project envisages that promoting soft, natural sea defences as the main form of flood defence will, in time, result in at least some of the hard defences becoming redundant and

either no longer needing to be repaired or being removed completely with natural dune vegetation reinstated in their place.

At Sea Palling in Norfolk, a seawall was constructed in the 1950s following a breach of the dunes in the 1953 flood. This severed the mobile (yellow) and fixed (grey) dunes from the embryo dunes and beach. As a result of falling beach levels, the Environment Agency constructed 8 nearshore, shore-parallel reefs to accrete sand. The resulting increase in beach levels went some way to offset the severance of the dunes from the beach and re-establish a supply of sand to the dune system. So although dune restoration was not an explicit objective of the intervention, it did provide some benefit.

5.3.2 Confidence in the science

Dune management needs to be based on a clear understanding of the physical and biological processes affecting the beach/dune system and how it will evolve (CIRIA 2010). This is often hampered by several causes of uncertainty including:

- future evolution of the shoreline under current forcing factors
- changes to current forcing factors (waves, currents, sea level rise)
- value of backshore to FCERM
- cyclical dune evolution, for example, dune accretion can occur over decades or centuries but significant erosion can occur in a single storm event

WWNP means accepting that erosion and accretion are both natural elements of dune evolution. Maintenance of natural evolution is likely to be preferable to costly and environmentally disruptive intervention. However, there may be short-term or long-term losses of backshore assets, natural habitat and built structures, which can lead to local interest groups calling for more positive management to be taken. Reducing uncertainties surrounding the spatial and temporal scales of dune evolution is therefore crucial to their successful FCERM role.

Confidence in future dune evolution may be affected by climate change, for example, rising sea levels, increased storminess and changes in precipitation affecting dune vegetation and hence stability.

More data are also needed on the FCERM standard provided by dunes and how it compares with other forms of flood defence in terms of performance and economics.

As with beaches, adaptive management is a way forward to addressing the uncertainties in WWNP in dunes and allowing sustainable long-term management while avoiding costly or environmentally disruptive engineering. To be effective, adaptive management needs to consider land use planning and environmental management over an area that may extend well beyond the dunes themselves. It also needs to take a long-term perspective, allowing for the cyclical nature of dune accretion and erosion. Adaptive management itself needs to take a cyclical approach, making small incremental changes based on observation and analysis, monitoring the results and then introducing further changes based on the outcome. Clearly, these approaches will not be applicable in all locations and their applicability depends on the assets at risk as well as the space available for dunes to evolve.

5.3.3 Metrics of cost–benefit and multiple benefits

Successful dune management has clear potential benefits to flood and coastal defence, nature conservation and recreational amenity.

Dunes contain a number of European Priority habitats and numerous rare species, particularly flora and invertebrates. These features are in general most effectively conserved when dunes are allowed to function naturally, with a significant degree of mobility. Long-term dune stabilisation results in ecological succession to grassland or scrub, with a reduction in characteristic dune habitats and species.

Dunes are also valuable recreational assets. However, there is also potential for recreational use to degrade the flood and coastal defence function of dunes and their nature conservation value. High levels of pedestrian traffic can erode vegetation, while vehicles such as dune buggies can be even more damaging. Erosion from recreational use can be managed, usually by fencing, boardwalks and signage.

The National Trust's South Milton Sands project demonstrated that it is possible to implement a scheme that addresses FCERM, nature conservation and public recreation in a heavily used area. However, following severe winter storm in 2013 to 2014, it is not clear that long-term FCERM benefits have yet been achieved.

There is a lack of quantified FCERM benefits from any dune management projects. Nor have any ecosystem services valuations relating to dunes been identified that would enable metrics for multiple benefits to be quantified.

5.3.4 Effectiveness/performance

Where they occur, sand dunes play a significant role in FCERM, acting as a natural line of defence against sea flooding. However, no UK examples have been found where dune restoration has been undertaken primarily as a quantified FCERM measure, rather than primarily as a conservation measure which may also have FCERM benefits. It is understood that such projects have been undertaken in the Netherlands.

To be effective in FCERM, there needs to be a balance between active dunes with a supply of sand and stabilised (vegetated) dunes. In the absence of a supply of sand to maintain active dunes, erosion of first active dunes and then fixed dunes is likely to occur.

The 5 main strategic options considered by Shoreline Management Plans (SMPs) can all be related to the management of coastal sand dunes for the purposes of coastal flood defence (Pye et al. 2007):

- **Do nothing**. Allow natural processes to take their course, possibly resulting in frontal dune erosion, blowout development and formation of transgressive dunes/sand sheets where the balance of forcing factors and human pressures favours this. Allow accretion of new dunes to occur where favoured by forcing factors and sediment supply.
- Hold the line or static preservation. Undertake engineering works (hard or soft) to maintain the position of the beach–dune interface and raise the level and width of the frontal dunes where necessary to ensure they can withstand the impact of a severe storm (for example, 1 in 200, 1 in 1,000 or 1 in 10,000 year event).
- **Dynamic preservation**. Maintain the general position of the coast but view the frontal dunes as a buffer zone within which limited change is permitted to take place during storms, and within which a degree of natural process instability is encouraged for the benefit of nature conservation.
- **Managed realignment**. Re-position the dune sea defence and create a second line of defence, inland of the frontal dune, using imported sand, reprofiling or existing sand and/or vegetation planting. Alternatively allow or encourage the

frontal dunes to roll back in a controlled manner to maintain the integrity of the defence, cease maintenance and possibly remove existing artificial defences and dune toe protection.

• Advance the line. Construct fences and plant vegetation to seaward of the existing foredune ridge. Encourage vegetation development in areas of natural potential accretion by beach nourishment and/or create artificial dunes by bulldozing, planting and fencing.

Wherever possible, coastal dune and beach systems should be allowed to respond naturally to changes in forcing factors and sediment supply conditions (Pye et al. 2007). Where accommodation space exists and conditions are favourable, frontal dunes should be allowed to roll back to establish a new equilibrium. However, in areas of low wind energy or strongly negative beach sediment budget, dune dissipation is likely to occur unless nourishment with fine-grained sand and artificial dune profiling are carried out. It is recommended that a detailed geomorphological evaluation study should be undertaken at each dune site, or group of sites, to assess the requirements and to identify the most appropriate management strategy. This will require nature conservation and other interests to be taken into account. Where not in existence, systematic monitoring programmes should be set up to provide early warning of dune change.

From the perspective of coastal flood defence, the most important issue relating to the hind dune areas concerns the degree of blowing sand and dune mobility allowed within the system (Pye et al. 2007). Traditionally, coastal engineers have considered it desirable to maintain a stable system, since this keeps a much sand as possible near the beach for as long as possible and prevents problems associated with sand invasion at the landward margin. However, the internal dynamics of a dune system have a large influence on the dune habitats that develop, and thus the range of vegetation communities and wildlife that exist. In general terms, a dynamic landscape favours biodiversity, although there can also be a risk that too much change may result in the destruction of particular ecological features that are of importance for nature conservation. A strategic decision therefore has to be taken, both for flood defence and nature conservation reasons, regarding the degree and spatial distribution of mobility/stability which is desirable. Following from this, decisions need to be taken about land use and access within the dune area, for example, the grazing regime, extent of afforestation, recreational activities and visitor management.

SEPA (2015) recommends that, where space exists, frontal dunes should be allowed to roll inland and establish a new equilibrium. In areas of low wind energy or a negative beach sediment budget, this is not likely to be possible so artificial dune profiling and restoration may be required. Generally, to reduce the risk of flooding, restoring sand dunes is primarily about stabilising or increasing the height, width or accretion rate of eroding dunes. Where current rates of erosion are between 1m and 10m per year, or dunes have been badly trampled, it is likely that immediate action is required. Where erosion is less than this, action may not be required as dunes could be in dynamic equilibrium (cyclically eroding and accreting), or the erosion could be insignificant. Under extreme storm conditions or a series of smaller events that occur close together in time, a dune system may be eroded altogether, overtopped or flattened. Wind scour of newly exposed sand can accentuate this. In these cases restoration may be appropriate to accelerate natural recovery processes and maintain levels of flood protection.

The following measures can be undertaken to help stabilise dunes (SEPA 2015):

• Fencing. This should be parallel to the dune face (shore parallel) with short spurs running towards the dunes (shore normal/perpendicular). It should be slightly forward of the toe of the dune. A void-to-solid ratio of 30–50% is recommended. Fencing helps prevent trampling or grazing, and can also reduce wind speeds to encourage the deposition of sand. Fencing can become an issue

for beach users over the course of time as the dunes accrete, for example, if the fencing becomes largely covered by accreting sand and presents a trip hazard.

- **Thatching**. This is the placement of timber or brushwood cuttings on the exposed dune surface to reduce wind speed and increase the deposition of sand. It is particularly effective where the amount of wind-blown sand is considerable. It is recommended that thatch should cover 20–30% of the exposed sand surface and should not be carried out on slopes with a gradient greater than 1 in 2. Beach users often remove thatching to make bonfires, so ongoing maintenance is required.
- **Planting vegetation**, typically marram grass, lyme grass or couch grass (*Elymus farctus*). It is best to transplant established dune grasses from a nearby site and is likely to take 2–3 years before transplants begin to thrive and spread. The vegetation and root network physically trap and hold sand in place, with organic matter and microorganisms also crucial to binding sand grains together. It is recommended that the vegetation should be planted at least 30m or more landward of mean high water.

Specific actions to engineer the dune landscape for biodiversity and conservation objectives also need to be considered. Examples include options to create artificial breaches, washover areas and shallow saline lagoons within the frontal dune area, or the reactivation of blowouts and recreation of wet slack areas in the hind dune area. A number of such experiments have been carried out in the Netherlands, England and Wales, with variable reported degrees of success. All such decisions should be based on a detailed understanding of the natural process regime and geomorphology of the dune system, as well as the flood defence and nature conservation requirements.

No examples have been found where the effectiveness of dune management as flood defence measures have been estimated or quantified.

5.3.5 Key case studies

This section outlines two UK and one Dutch experience with sand dune restoration for FCERM. Examples elsewhere in the world are documented by Martinez et al. (2013).

South Milton Sands, Devon

South Milton Sands is a heavily used 4ha sand dune site with a small beach and extensive car parking. It is popular for beach enjoyment and swimming. In 2002, the wooden piling defences constructed in 1990 were at the end of their lifespan, not working properly and thought to be unsustainable in the longer term in the context of sea level rise. Following options appraisal and extensive consultation, the defences were removed (including rotten timber piling and many tonnes of rubble) and the dunes were reprofiled. Once groundwork was complete, local people helped plant marram grass on the dunes (CIRIA 2010).

The National Trust wanted to keep the beach and dunes for recreational use rather than as a car park, to implement WWNP and to engage the local community. The public was concerned about loss of car parking and erosion of the access track. It was agreed that a small area of wooden piling would be retained for 10 years, covering the slipway area and access track. This gave people time to accept that change will occur over time during the transition from defended to natural dunes. A small parking area was retained onsite with good views out to sea and access to the beach was improved with boardwalks (CIRIA 2010).

The project took 6 years to complete, mainly because of the stakeholder engagement process. People need time to understand and come to terms with change. Engaging the

community in practical tasks of dune restoration generated positive publicity and helped the project being viewed as a success (National Trust, undated).

The National Trust reported that 'the dune replanting programme was a great community effort and well-meaning' (Tony Flux, personal communication). However, it proved shortlived as much of the restored dunes were lost in a major storm on 14 February 2014. Storms and high tides significantly damaged the dune system and the access track that lies behind it. Significant erosion to the dunes occurred, plus damage to pedestrian access routes and a National Trust slipway, and the loss of the eastern 50m of an unpaved roadway. The Trust reinstated these features, diverting the road inland. However, restoration of the dunes and marram planting was not attempted due to the significant loss of sand, beach lowering and dune erosion.

South Milton Sands beach has since recovered quite well in terms of beach levels, but only a little embryo dune reformation has occurred. While the frontal dunes acted as a reasonable sea defence, there were no secondary dunes behind to take their place when the frontage was washed out. Tony Flux, the National Trust's Coastal and Marine Adviser for the southwest, concluded:

Should the remainder of the frontal dunes be removed in a future storm (which is a very high probability), we will have to roll back the service road a second time and probably to a more durable location but there is no question of either trying to rebuild the dunes or install artificial defences. The marram grass planting exercise was all done in good faith, but on reflection was never really going to make a sustainable difference to the durability of the dunes or their profiles'.

A location for the dunes further landward might be more sustainable.

National Trust does not have any data on the FCERM standard at South Milton Sands before the dune restoration, following dune restoration or following the 2014 storms. The 14 February 2014 storm was estimated at a 1 in 250 year standard, and partly overtopped but did not fully breach the dunes.

Beach profile monitoring over a 700m frontage at South Milton Sands showed a loss of 4,500m³ over the period 2007 to 2013, and a further loss of 13,700m³ during the winter 2013 to 2014 storms (Plymouth Coastal Observatory 2014). No recovery had been noted by summer 2014 and indeed beach volumes had fallen further. The total estimated beach volume on 11 August 2014 was 284,815m³ compared with 288,115m³ on 5 March 2014 and 301,799m³ on 18 October 2013 (Plymouth Coastal Observatory 2014). Recovery of the beach level had occurred between 2014 and 2016, and a site visit in December 2016 noted that some embryo dunes were starting to reform. The standard of defence against overtopping or breaching has not been estimated, but a further major storm is likely to breach what remains of the dunes.

Hightown Dune Restoration Project, Sefton, Merseyside

Sefton Council undertook a dune restoration project in 2011 to provide improved FCERM to 125 properties at Broseley housing development at Hightown. The objective was to reinstate sand dunes along the Hightown frontage to the position they were in during the late 1970s. In addition, the seawall at Blundellsands Sailing Club was replaced with a sloping revetment and armouring provided to a nearby outfall to keep sand in place.

Sand was sourced from behind the promenade at Crosby from the Marine Lake up to Crosby Swimming Baths. A quantity of 27,000m³ of sand was transported by large off-road trucks along the beach and landward of the beach (not on public roads). The sand was placed seaward of the existing sand dunes and management measures applied to stabilise the dunes, such as marram grass planting.

The works cost £1.4 million, funded from money provided by the developer of the Broseley Housing Estate set aside for this purpose in the 1970s. Environmental issues included loss of sand dunes at Crosby and disturbance to birds on the beach at high tide. A project appraisal report (Sefton Council 2011) and Environmental Statement were produced.

A year after the project was implemented, large storms had eroded parts of the dunes and sand had been redistributed to the beach, raising beach levels. This is part of the natural adjustments of the system to the new position of the dunes (Sefton Council 2012). The project was considered successful, with the dunes and beach acting as predicted. However, Graham Limbrey of Sefton Council (personal communication, 2017) stated:

'We still struggle to understand the long-term benefits of sand dune management. The techniques are well documented from an operational perspective and they have been demonstrated to have a short-term benefit (that is, months) but there is no evidence of the benefits that occur over a number of years'.

Ameland, Netherlands

De Jong et al (2014) described the evolution of Dutch thinking about dune management since the 1980s. The idea gained currency that drifting sand is necessary to preserve the natural character of coastal landscapes. In 1990 this notion was embraced in the Netherlands' first policy document on coastal management. A major driver of this shift in thinking was the occurrence of numerous storm surge events in the late 1980s. Moreover, continued erosion was measured in a number of places. This proved false the earlier assumption that the sum total of erosion and accretion along the Dutch coast was zero.

The Dutch government committed itself to stopping any further coastal recession. It established a 'reference' coastline (*basiskustlijn*), which was to be maintained at its 1990 position using nourishments. Recharge is typically achieved by depositing sand on a beach, on a shore face, or in front of a foredune ridge. Initially most nourishments were done on beaches, though later insights (from 1997) led to more sand placement on the shore face (at about the 5–6m isobath). Since 2000, some 12 million m³ of sand have been added annually to the Dutch coast, compared with about 6 million m³ per year before 2000.

De Jong et al (2014) presented a case study on dune flood defence on the North Sea barrier island of Ameland in the Netherlands. Two types of coastal dune management are reported:

- a soft engineering approach, in which sand fences are placed on the seaward side of foredunes
- dynamic coastal management, with minimal or no dune maintenance

Measurements of the dunes showed that implementation of dynamic coastal management did not directly affect the volume of the foredune. Growth was occasionally interrupted, coinciding with high water events. In periods between erosive storms, dune growth rates did not show a significant difference between management types. The main effect of the change was on vegetation development. De Jong et al. (2014) concluded that introducing dynamic coastal management (that is, the absence of sand fences) did not reduce coastal safety.

5.3.6 Funding

Dune restoration for FCERM can be funded by Defra grant-in-aid. This was the case for the Sea Palling project conducted by the Environment Agency, though the objective here was beach management by constructing offshore reefs to raise beach levels and not dune restoration.

Most dune restoration projects are funded primarily for ecological restoration. The South Milton Sands project, for example, cost £150,000 and was funded by the National Trust, with a small contribution from the local Area of Outstanding Natural Beauty. Work on restoration of Welsh dune systems, primarily as a nature conservation measure, has been funded by Natural Resources Wales. Work on the restoration of Lancashire dune systems has been funded by the Fylde Sand Dune Project, whose steering group comprises local authorities, Natural England and NGOs.

Another potential funding mechanism is provided by Defra's Countryside Stewardship. This can pay capital and management costs in relation to eligible habitat creation, including:

- CT2: Creation of coastal sand dunes and vegetated shingle on arable land and improved grassland
- CT1: Management of coastal sand dunes and vegetated shingle

Although public authorities are not eligible to claim Countryside Stewardship, tenant farmers and NGOs such as Wildlife Trusts are able to do so for land that is leased to them.

5.3.7 Design, management and maintenance

Dune management is complex. There are forcing factors from the physical environment including long-term trends, cyclical trends and storm events, processes of ecological succession and frequently anthropogenic pressures from recreational use. Typical measures in a dune management plan (CIRIA 2010) include:

- permanent and seasonal fencing
- planting to stabilise bare sand areas
- educational signage to improve awareness of the dune system
- liaison with beach users to improve awareness of the dune system
- boardwalks to reduce trampling
- · zoning schemes to control access to vulnerable areas
- providing designated barbecue areas
- localised protection of infrastructure
- regular monitoring

At Formby in Lancashire and Porthtowan Sands in Cornwall, old Christmas trees have been used as a stabilisation technique on the dunes and to encourage sand accretion.

At South Milton Sands, the National Trust has instigated the following management measures since capital works were completed, mostly in response to storm damage (Tony Flux, personal communication):

- marram planting by community volunteers
- repairs to address undermining of slipway
- extending wooden palisade to reduce outflanking risk
- reinstatement of an access track serving a café
- temporary signage to request public to keep off eroding dune faces

In a report on Welsh dunes, Pye and Blott (2012) recommended rejuvenation intervention options ranging from increased stock grazing to much larger scale intervention measures including:

- scrub clearance
- stripping of areas of fixed grassland vegetation
- creation of artificial trough blowouts in the frontal dunes
- · excavation of artificial 'blowouts' in inland dune areas
- localised placement of excavated sand to enhance local wind flow
- beach and dune nourishment
- removal of artificial features which impede the operation of natural processes, such as conifer plantations

At Fylde Dunes in Lancashire, Skelcher (2008) recommended management works to increase the area of wildlife habitat and to improve the efficiency of flood defence by:

- enabling natural seaward accretion of the dunes by removing the current causes of human-induced erosion)
- grassland and scrub management works to enhance the nature conservation value of the existing dunes

5.3.8 Other requirements

Most significant dune systems are protected as Sites of Special Scientific Interest (SSSIs) and Special Areas of Conservation (SACs). Management measures will therefore require agreement from nature conservation bodies such as Natural England and Natural Resources Wales. Planning consent from local authorities may also be required.

5.3.9 Research gaps

Sand dunes are a valuable natural sea defence but there are few, if any, UK examples of successful dune restoration primarily for quantified FCERM benefits. Dune restoration projects have been driven primarily by nature conservation enhancements and the need to manage recreational impacts.

Obstacles to dune management for FCERM include both the long-term cyclical nature of natural dune evolution and the need for extensive accommodation space for dunes to roll back and realign. The spatial and temporal scales involved in these processes are frequently at odds with objectives to provide short-term flood risk management to built assets in the immediate hinterland of the dunes. Where modest-scale dune management measures to provide FCERM, as well as nature conservation and recreation benefits, has been undertaken (for example, at South Milton Sands in Devon), the FCERM benefits appear to have been short-lived.

Conversely, historical forms of management aiming at stabilising dunes (for example, by tree planting) are inconsistent with nature conservation objectives and are unlikely to provide sustainable FCERM in the context of rising sea levels.

There may be scope to identify and develop a FCERM demonstration project on dune management. To have a realistic chance of successful delivery, selection of a suitable large-scale site with sufficient accommodation space would be crucial.

Important research gaps include:

- predicting future evolution and changes to the shoreline under current forcing factors (waves, currents, sea level rise)
- value of dune backshore to FCERM (estimating defence standard)
- predicting cyclical dune evolution (for example, dune accretion can occur over decades or centuries but significant erosion can occur in a single storm event)
- predicting sand supply at individual dune sites
- research into the long-term benefits of sand dune management (while the techniques are well-documented from an operational perspective and they have been demonstrated to have a short-term benefit, that is, months, there is no evidence of the benefits that occur over a number of years)
- implementation of a large-scale FCERM dune management project in UK

5.4 Managed realignment and regulated tidal exchange

5.4.1 Understanding the science

Managed realignment is defined in SMP guidance (Defra 2006) as:

'allowing the shoreline to move backwards or forwards, with management to control or limit movement (such as reducing erosion or building new defences on the landward side of the original defences).'

In Britain, managed realignment has usually involved the deliberate removal of an existing line of coastal defence and its substitution with either a natural or maintained line of defence further landward. In this more limited sense, managed realignment is distinct from limited intervention, whereby the natural process of shoreline retreat is managed, as for example in the case of cliffs or dunes. This section deals with the deliberate removal of defences and substitution with a new defence further landward.

Over the past 20 years, managed realignment has become a well-established approach to coastal management, mainly within estuaries but also on the open coast (for example, at Medmerry in West Sussex). The principal drivers for managed realignment include both FCERM and biodiversity (often driven by the Habitats Directive). They include:

- improving FCERM through providing a higher standard of service and/or reducing the costs of maintaining realigned defence
- creating a more natural or shorter shoreline and achieving greater sustainability of realigned defence
- creation of new, restored replacement or compensatory intertidal habitat such as saltmarsh, with associated ecosystem services
- increased accommodation space or flood storage within estuary environments leading to a reduction in extreme water levels

Other objectives cited by managed realignment schemes in Britain and elsewhere in western Europe (OMReg) include mitigating climate change, improving water quality, creating a recreational facility and acting as a demonstration/pilot scheme.

The science behind managed realignment is generally well-understood as a result of extensive experience implementing managed realignment at a wide variety of different scales and locations. Useful reference sources describing the principles and practicalities are CIRIA (2004) and RSPB (2005).

Managed realignment is usually undertaken as either 'breach' realignment (where a discrete breach is made in the existing defence) or 'bank' realignment (where the whole existing defence is removed). Bank realignment is more expensive, but may be better suited where the objective is to provide a wide, dissipative intertidal surface.

A related coastal management approach is regulated tidal exchange, where intertidal habitat is created landward of a seawall without making a full breach in the existing defence. Usually, a sluice would be used to convey water in and out of the intertidal habitat, enabling the frequency and quantity of inundation to be more closely controlled. The potential advantages of this are that:

- it may enable saltmarsh to form on land that is too low in the tidal frame to sustain saltmarsh if managed realignment were adopted
- where freshwater input is available, it could be used to create brackish rather than fully saline wetland habitat
- it may avoid the need to invest in constructing or raising a retreated line of defence

Because regulated tidal exchange requires the existing defence line to be maintained, it is not itself a FCERM measure. However, it may have a role in FCERM if it is adopted as an interim measure to raise the level of the inundated area through natural sediment accretion, as a prelude to full managed realignment at some point in the future.

5.4.2 Confidence in the science

This section outlines potential uncertainties relating to confidence in the underlying science and effectiveness in fulfilling FCERM and other scheme objectives through managed realignment.

Intertidal habitat creation, restoration and compensation

Based on an analysis of the OMReg database, biodiversity gain or compensation is cited as an objective of 95% of managed realignment schemes implemented in the UK.

The broad type of intertidal habitat that forms in an area of managed realignment is principally a function of topographic elevation and consequent frequency/duration of tidal inundation. These factors are well-understood and can be predicted in advance using tidal curves and in more detail by modelling.

Other physical factors such as substrate type, slope, salinity and exposure to currents and waves also play a role. These are less readily quantified. In many cases, the best guide to the type of habitat that will form is to examine the types of habitat already in existence in adjacent intertidal areas at similar elevation.

Often the objective is to create saltmarsh, on account of its intrinsic habitat value and to replace losses of saltmarsh elsewhere. Saltmarsh can also have an FCERM function in attenuating waves and therefore reducing the size of a realigned defence, though this does not appear to have been a primary driver in many cases. In general, saltmarsh forms in intertidal areas between mean high water neap and mean high water spring levels, where the intertidal surface receives up to 300 inundations per year. Pioneer marsh is likely to form

where there are between 300 and 450 inundations per year (RSPB 2005). Clay or clayey loam soils are considered to facilitate saltmarsh development more rapidly than gravel, sandy or alluvial soils, though all soil types can lead to saltmarsh formation. Wave action above a critical level will erode saltmarsh.

Sometimes the objective may be to create mudflat as a compensatory habitat measure. Mudflat usually forms below mean high water neap level, where the intertidal surface receives more than 450 inundations per year.

It is generally possible to predict the type of habitat that will form on a given managed realignment site on the basis of its physical characteristics and comparison with intertidal habitat in adjacent areas. Comparison can be done using digital topographical height data (for example, remotely sensed using LiDAR imagery) overlaid with habitat maps to find the 5th and 95th percentiles of elevation at which a particular habitat such as saltmarsh (or one of its constituent vegetation types) is present (CIRIA 2004). These give a reasonable estimate of lower and upper bound elevations for that habitat. However, experience has found that some new intertidal sites do not form the expected habitat, for example, because the substrate density or chemistry may be unsuitable. This is an area where more research into the causes may increase understanding and hence the predictability of habitat creation.

Improvements to flood risk management and reduced defence costs

In theory, managed realignment can reduce the size of realigned defence required compared with holding the line, through the contribution to attenuating waves made by intertidal habitat within the realigned area. However, realigned defences may be higher than those they are replacing to provide an increased defence standard or cater for projections of future sea level rise.

Defence design is well-understood using standard techniques for predicting water level return periods and wave overtopping. In the case of saltmarsh, the principal variables affecting wave dissipation are considered to be (CIRIA 2004):

- height of the wave approaching the marsh
- width of the marsh through which the wave propagates
- water depth
- size of the plants (stem or foliage diameter)

The effectiveness of saltmarsh as a contribution to coastal flood risk management is discussed in later. A number of theoretical and practical studies have shown that saltmarsh ecosystems can be a valuable component of coastal protection schemes.

However, the potential savings in defence construction as a result of having saltmarsh to the seaward are less easy to realise in practice. Firstly, there may be uncertainties relating to the new habitat that will form and therefore its effectiveness in attenuating waves. Secondly, defences are typically designed for a period of 100 years (including predicted sea level rise) and it is difficult to be confident that saltmarsh will still be present many decades ahead, given the effects of rising sea levels and wave erosion. Defence design is necessarily conservative and the prudent approach is to assume that saltmarsh will not last for its design life. The theoretical savings may therefore be difficult to achieve. One way in which savings could be achieved through deferring costs would be to design defences for a shorter life assuming saltmarsh is present, with a view to revisiting the design in a future years and where necessary raising the defences. For this to be workable, foundations for seawalls need to be specified for their potential future height. This is an area where refinements to existing guidance on defence design and on the allocation of risk within design and

implementation may be needed if theoretical savings in realigned defence construction are to be achieved.

In the UK, one major managed realignment project has been implemented on the open coast with the primary objective of improving FCERM is at Medmerry in West Sussex. This is covered in more detail as one of the project case studies.

Estuary hydraulics

Managed realignment in an estuary will increase its tidal prism and may affect water levels elsewhere in the estuary. The provision of flood storage or accommodation space is an objective of some schemes, usually where there is a fluvial component to flooding or the scheme is located towards the landward end of the estuary and water levels are hydraulically restricted.

Increase in the tidal prism may increase erosion of defences elsewhere in an estuary, particularly downstream of the realignment site. The tidal prism is the volume of water exchanged through a coastal or transitional system, typically measured between mean low water spring and mean high water spring levels. Changes in tidal prism and currents can be predicted by modelling using the dimensions of the site, tidal regime and hydraulic characteristics of the estuary. The effect of such changes on erosion of existing defences requires expert engineering judgement. These considerations are important as some schemes have led to significant downstream erosion within a few days of opening.

Depending on its location, managed realignment may increase or decrease water levels within an estuary, with consequent negative or positive effects on flood risk management elsewhere. Predicting this is a complex issue requiring numerical modelling. Previous modelling work on the Humber Estuary found that managed realignment schemes located in the inner reaches of an estuary may generate reductions in peak water levels, whereas schemes in the outer reaches may result in slight increases (Townend and Pethick 2002).

Large realignment sites near the mouth of an estuary have the potential to raise water levels throughout the estuary. This increase is produced by the newly created intertidal area drawing additional water into the estuary on the flood tide. Although some of this additional water enters the realignment site, a proportion also bypasses the realignment site and continues upstream, thus increasing water levels in these areas of the estuary. Modelling suggests that an important determinant on estuary water levels is the rate at which the realignment site fills. Schemes that fill more slowly are predicted to have less impact than those that fill more rapidly, even if high water levels within the schemes are similar (Pontee 2015).

Creating a more natural shoreline

Managed realignment usually involves recreating intertidal habitat on land that was formerly reclaimed from the sea. To that extent it represents the restoration of a more 'natural' shoreline. However, if a new line of defence is constructed to landward, as is frequently the case, then the shoreline will still be to some extent artificial.

Climate change

Managed realignment is often intended as a response to future climate change. In the sense that realigned defences are designed for predicted future sea level, this is true of nearly all such schemes. Most schemes make estuaries and coastlines more resilient to increased sea

levels by accommodating the landward shift in shorelines that would occur if defences were not maintained and thus WWNP.

Some managed realignment schemes go further and are intended to mitigate the **effects** of climate change by reducing predicted future increases in water levels within estuaries. Whether managed realignment will increase or decrease water levels and hence increase or decrease stress on existing defences depends on its location within the estuary and the overall dynamics of the system. For example, prior to its construction, the Environment Agency predicted that the Alkborough managed realignment scheme in the upper Humber would reduce high tide levels over a large part of the upper estuary by 100mm to 150 mm. At a projected annual sea level rise of 4mm per year until 2025, and then 8.5mm per year until 2055, the Alkborough scheme would therefore modify the estuary regime to account for perhaps 25 years of this climate change impact (Environment Agency 2009). In general, the creation of accommodation space to reduce water levels within estuaries through managed realignment makes them more resilient to climate change, reduces stress on existing and realigned defences, and increases sustainability in the context of rising sea levels.

Some schemes include within their objectives mitigating the **causes** of climate change through carbon sequestration. Prior to its construction, Eftec (2007) estimated that the Alkborough managed realignment scheme, if implemented with a 20m breach, would sequester 539 tonnes of carbon per year. If the breach were increased to 250m, the additional intertidal area would result in 770 tonnes of carbon per year being sequestered. However, these figures need to be set against the estimate of 550 tonnes of carbon per year absorbed by the site if no managed realignment were implemented. It is likely that the realisation of a benefit from managed realignment depends on the existing and alternative land uses of the site. Some uses, such as biomass crops or timber production, would be likely to achieve higher carbon sequestration rates than managed realignment.

Natural England gathered together some evidence on carbon sequestration (Natural England 2015). A study of the Blackwater Estuary in Essex looked at how managed realignment sites might affect carbon storage and greenhouse gases. Potentially, managed realignment sites in the estuary (29.5km² of saltmarsh and 23.7km² of intertidal mudflat) could sequester 5,478 tonnes of carbon per year. However, greenhouse gas emissions of methane and nitrous oxide would reduce the net benefit by 24% to 4,174 tonnes of carbon per year. A similar calculation applied to the Humber Estuary suggested that adding 7,494ha of new intertidal area to the estuary would lead to an annual accumulation of 1.2×10^5 tonnes of new sediment, increasing the carbon sink potential of the estuary by 150%. This figure is offset by greenhouse gas emissions, reducing the potential benefit by over 50%. Creating about 25% of the maximum potential area for managed realignment on the Humber Estuary (26km² of land) would potentially store 40,000 tonnes of sediment per year, thereby burying around 800 tonnes of organic carbon. A comparison of agricultural land, natural saltmarsh and management realignment sites in Essex found that soil carbon stock, carbon to nitrogen ratio and below-ground biomass in managed realignment sites were more similar to agricultural land than natural saltmarsh. This study suggests that the carbon storage potential of managed realignment sites may take 100 years to reach the full potential of the natural sites at storing 0.92 tonnes of carbon per hectare.

Water quality

Although no UK schemes cite improving water quality as an objective, one scheme in Germany and one in the Netherlands included this among their objectives (OMReg). There is no information available in the UK about the success of these schemes in meeting this objective. Environment Agency (2009) notes that reedbeds and mudflats are efficient water purification habitats. Their roles include nitrogen stripping and reducing biochemical oxygen demand. Saltmarsh is also effective at encouraging sedimentation of material suspended in

the water column, leading to reduced turbidity and greater water clarity. However, Environment Agency (2009) considers that there is insufficient evidence available to quantify improvements to water quality resulting from managed realignment. This is an area where further research may improve confidence in realisable benefits.

Regulated tidal exchange

Regulated tidal exchange allows tidal inundation to be controlled to a defined degree. It can either be used in its own right or as a precursor to future bank or breach realignments. Regulated tidal exchange is particularly suited to sites where the environmental impacts of unconstrained inundation are uncertain, saltmarsh vegetation is desired but the site is too low in the tidal frame, or where existing flood defences cannot be removed (CIRIA 2004).

As a precursor to bank or breach managed realignment, regulated tidal exchange represents an FCERM measure, particularly where used to accrete sediment. This may take many years to achieve, especially where sediment supply is limited.

Another example of regulated tidal exchange indirectly linked to FCERM objectives is the scheme at Rye Harbour Farm in East Sussex (Environment Agency 2016b). The farm was purchased in 2003 as part of a project to upgrade sea defences along the Pett frontage. Material to construct a realigned defence bank was sourced from Rye Harbour Farm. The borrow pits were subsequently remodelled to create a creek network. This allowed the site to be restored to saltmarsh using regulated tidal exchange, and so biodiversity gain was provided as an add-on to the FCERM project.

Regulated tidal exchange can be achieved by lowering the crest of defences to create a spillway or by culverts or pipes through the defence. Tidal flow is restricted by the invert level and sizing of these systems and may be further regulated by penstocks or sluice (flap) valves. The relatively small hydraulic capacity of spillways, culverts and pipes compared with bank or breach realignment usually restricts their use to sizes of a few hectares in size, though regulated tidal exchange schemes have been implemented in Germany and the Netherlands to control flooding over more extensive areas.

5.4.3 Metrics of cost–benefit and multiple benefits

Environment Agency (2015f) provides estimates of the costs of managed realignment in different parts of England. ABPmer (2015) notes that the cost of managed realignment is influenced by a number of factors, including:

- extent of site manipulation/engineering works required to achieve the underlying objectives
- location and desirability of the land
- efforts involved in gaining planning consent

ABPmer's review of the implementation costs for completed UK schemes has shown that unit costs (that is, cost per hectare) can vary widely, depending very much on the location, engineering effort and objective of a given scheme.

The lowest per hectare costs have been reported for the Lantern Marsh scheme on Orford Ness (Suffolk) with just under £800 per hectare, whereas the most costly scheme to date has been the Trimley scheme on the Orwell (Suffolk) at almost £123,000 per hectare. Such variability is to be expected given the distinct challenges and constraints faced at the individual schemes. Over the past 25 years, unit costs of managed realignment schemes (excluding regulated tidal exchange schemes) have averaged just over £35,000 per hectare

(2014 prices, n = 291, excluding one outlier). Since 2000, there has been a slight upward trend in average costs, with post-1999 schemes averaging just over £40,000 per hectare (n = 23, excluding one outlier).

The economic benefits of managed realignment compared with 'hold the line' may include:

- reduced whole life cost of constructing and maintaining flood risk management defences
- ecosystem services provided by created habitat such as food production (sheep and cattle grazing, fish), wool, water regulation, carbon sequestration, storm protection, recreation and visual amenity

The economic costs of managed realignment include:

- value of land no longer defended (usually agricultural land)
- loss of ecosystem services provided by land no longer defended, such as food production (arable or pastoral), recreation and visual amenity

Environment Agency (2013a) considered the quantifiable benefits from managed realignment schemes in the Humber Estuary to arise in 6 ecosystem services categories:

- food and raw materials
- coastal protection
- habitat
- nutrient cycling and waste treatment
- climate regulation
- recreation, cultural and aesthetic services

Managed realignment schemes are usually promoted on the basis of multiple benefits. Most UK schemes include both improved flood risk management and biodiversity gains among their benefits. Many are also justified principally or partly on the basis of creating compensatory habitat (that is, offsetting losses of intertidal habitat elsewhere to comply with the Habitats Regulations). At least one UK scheme (Alkborough) was promoted in order to increase accommodation space to reduce extreme water levels within an estuary.

A number of studies have been conducted to value cost–benefits from managed realignment projects such as at Alkborough (Environment Agency 2009) and at the Steart Peninsula in Somerset (da Sivla 2012 and da Silva et al. 2014). Eftec (2007) examined Alkborough (Humber) and Wareham (Poole Harbour) as theoretical case studies, prior to their implementation. These studies all use values for ecosystem services derived from other studies reported in literature, some of which are subject to substantial uncertainties. Environment Agency (2009) valued the coastal defence function at Alkborough Flats to be worth £12.26 million over a 100 year time frame. No such values were found for the other managed realignment sites in the Humber.

A common view is that managed realignment may lead to some loss of 'provisioning' services (for example, food and fibre production). However, this is more than offset by a gain in 'regulatory' services (flood risk management), 'supporting' services (biodiversity) and 'cultural' services (for example, amenity). The conclusions of the Alkborough and Steart ecosystems services valuations are summarised below.

Alkborough Flats

The Alkborough managed realignment scheme in the Humber Estuary is estimated to have cost ~£10 million to build and to have provided £12 million of storm protection benefits to land and property (Environment Agency 2009). Other ecosystems services benefits linked to the scheme were ~£1 million per year (Table 5.3). The gross benefits of the Alkborough Flats scheme are therefore estimated at £27,989,899 made up of:

- gross benefits of natural hazard regulation (that is, storm protection) = £12,260,000
- total of annualised benefits for all other ecosystem service benefits = £933,726

Assessed over 25 years with a discount rate of 3.5%, this equates to a gross benefit for all other ecosystem services of £15,729,899. This then yields a gross benefit value of £12,260,000 ('natural hazard regulation') + £15,729,899 (all other ecosystem services evaluated) = £27,989,899.

The gross costs of the Alkborough Flats scheme are documented in the project appraisal report (Environment Agency 2005) as 'the present value of the cost of developing Alkborough Flats is £8.69 million'. This document records 'an average benefit to cost ratio of 2.72' based on more generalised habitat values. However, on the basis of the full suite of ecosystem services, the Alkborough Flats scheme yields an enhanced benefit to cost ratio of 3.22. Note that this is based on very conservative valuations.

Ecosystem service	Annual benefit assessed	Research gap/note
Provisioning services		
Freshwater	No net value ascribed as brackish site	
Food (for example, crops, fruit, fish)	-£28,075	Contribution of saltmarsh to fish recruitment
Fibre and fuel (for example, timber, wool)	£26,820 (wool minus straw)	
Genetic resources (used for crop/stock breeding and biotechnology)	£3,000	
Biochemicals, natural medicines, pharmaceuticals	No net value ascribed	
Ornamental resources (for example, shells, flowers)	No net value ascribed	
Regulatory services	·	
Air quality regulation	Not possible to quantify at present	Major research gap
Climate regulation (local temperature/precipitation, greenhouse gas sequestration and so on)	£14,553 from carbon sequestration	 microclimate assessment confounding greenhouse gas impacts

Table 5.3 Ecosystem services valuation of Alkborough Flats managed realignment scheme

Ecosystem service	Annual benefit assessed	Research gap/note
Water regulation (timing and scale of run-off, flooding and so on)	No benefit assessed	
Natural hazard regulation (that is, storm protection)	£12.26 million over 100 years at variable discount	
Pest regulation	No value ascribed	
Disease regulation	Neutral impact of scheme	
Erosion regulation	No value ascribed	Contribution from site to catchment erosion risk
Water purification and waste treatment	No value ascribed	
Pollination	No value ascribed	
Cultural services	'	
Cultural heritage	No monetary value assigned	
Recreation and tourism	£164,830 ignoring informal recreation	
Aesthetic value	No monetary value assigned	
Spiritual and religious value	No monetary value assigned	
Inspiration of art, folklore, architecture and so on	No monetary value assigned	
Social relations (for example, fishing, grazing or cropping communities)	No monetary value assigned	
Addendum services: navigation	Annual net cost of £5,000	
Supporting services		· · · · · · · · · · · · · · · · · · ·
Soil formation	Benefit not quantified	More direct measure of soil formation
Primary production	£8,160 (monoculture to complex habitat)	Quantification of secondary production
Nutrient cycling	Benefit not quantified	Quantification of nutrient cycling
Water recycling	Benefit not quantified	Quantification of water recycling
Photosynthesis (production of atmospheric oxygen)	No value assigned	
Provision of habitat	£749,438	
Addendum services: increased estuarine resilience	No value assigned	

Source: Environment Agency (2009)

Steart Coastal Management Scheme

In her Master's thesis, Da Silva (2012) estimated that the net annual benefit (including both monetary and semi-quantified gains and losses) of wetland habitat creation by the Steart Coastal Management project in Somerset was in the range £491,000 to £914,000. This is made up from the elements summarised in Table 5.4.

Ecosystem service	Annual benefit/cost assessed	Research gaps
Provisioning services		
Food	Loss of £37,300 to benefit of £83,850	Contribution of intertidal habitats to fish and shellfish biological productivity as well as Salicornia productivity.
Regulating services		<u>.</u>
Climate regulation	Benefit = £15,375–£46,125	
Water regulation	Not assessed to avoid double- counting. Improved hydrology for fish and wildlife to be assessed under 'recreation' and 'habitat provision'	
Water purification and waste treatment	Not assessed to avoid double- counting. Improved water quality for fish and wildlife assessed under 'recreation' and 'habitat provision'	
Cultural services	'	'
Recreation and tourism	Benefit = £300,840-£469,310	Recreational fishing evidence
Education	Benefit = £87,000–£132,000	Contribution of ecosystem services to tertiary education; intrinsic value studies of education (rather than using the cost-based approach)
Supporting services	'	
Soil formation	Benefit quantified but not valued	Lack of valuation studies in the literature
Primary production	Benefit not quantified	Lack of quantification methods in the literature
Nutrient cycling	Benefit not quantified	Lack of geochemical data
Water cycling	Benefit not quantified	Lack of local hydrological modelling data
Photosynthesis	Benefit not quantified Service linked to primary production	Lack of quantification methods in the literature
Provision of habitat	Benefit = £125,240–£182,467	Lack of valuation studies on biodiversity in the UK

Table 5.4 Ecosystem services valuation of Steart Managed Realignment Scheme

Source: da Silva et al. (2014)

5.4.4 Effectiveness/performance

The effectiveness of managed realignment as a FCERM measure depends on factors such as follows.

- Whether a constructed new defence is required. If realignment is to natural topography, it may not be, in which case the new line of defence would be inherently more sustainable.
- If a constructed new defence is required, the elevation of its toe relative to that of the seaward line of defence. If the new line is located on higher land its construction and maintenance costs would be lower and the new line of defence would be more sustainable.
- If a constructed new defence is required, its length relative to the seaward line of defence. If the new line is shorter in length, its construction and maintenance costs would be lower and the new line of defence would be more sustainable.
- Exposure of the new line of defence to erosion and overtopping by waves and currents compared with the seaward line of defence. If intertidal habitat such as saltmarsh is created in the realigned area, it would reduce exposure to waves. This may enable a lower crest height to achieve the same overtopping parameters or a reduced need for armouring to protect against scour. However, as described above, because defence design is necessarily conservative, cost savings in defence construction from these benefits can be hard to realise.

In general, managed realignment schemes are intended to increase the sustainability of FCERM by resulting in a line of new defence that is protected by intertidal habitat on its seaward side. Ideally, the new line of defence would be lower and shorter than for a Hold the Line option, but in many cases this has not been achieved. It would, however, be true to say that managed realignment usually results in an improved standard of defence compared with the existing situation and an alignment that is more sustainable. At Medmerry, for example, the standard of defence was as low as 1 in 1 year but managed realignment has increased it to a 1 in 100 year standard in year 100, protecting 348 properties, albeit by constructing 6.8km of new embankments (Environment Agency 2010b, 2010c).

Maintenance costs of realigned defences are typically lower than those of the original defences and often have lower construction costs than refurbishing the original defences in response to rising sea levels (Adaptation Sub-Committee 2013). The Environment Agency states that maintenance costs savings may be modest due to additional costs, such as new pumping stations, the need to maintain the breach in the old seawall, and in some cases a continued need to maintain the old wall. Reduced capital costs are also not automatic, as new defences tend to have wider crests and shallower. Analysis by the Adaptation Sub-Committee (2013) showed that these factors can considerably affect the cost-effectiveness of the managed realignment schemes, especially for smaller schemes that create less habitat area per kilometre of defence.

Inevitably, managed realignment is expensive compared with No Active Intervention.

It would be useful to further research the design parameters of UK managed realignment schemes to assess the extent to which lower, shorter and more sustainable defences have been achieved and the whole life costs of realignment compared with the alternatives such as Hold the Line.

Tinch and Ledoux (2006) reviewed the economic case presented for 6 UK managed realignment schemes (Orplands, Freiston, Abbotts Hall, Paull Holme Strays, Brancaster and Nigg Bay). They concluded that differences in the assumptions and format, together with site-specific considerations, meant it was not possible to draw general conclusions about the

economics of managed realignment compared with other options. General points they made are as follows.

- Managed realignment schemes often have lower maintenance costs than other alternatives. This would be expected as a function of managed realignment schemes involving lower embankments located further inland.
- The cost per hectare of habitat created varied over 2 orders of magnitude from just £1,500 at Nigg Bay to £125,000 at Freiston (30ha option) or £90,000 at Brancaster (most expensive actual scheme).
- The cost-effectiveness of managed realignment compared with other options from a flood defence perspective is difficult to demonstrate owing to inconsistencies in appraisal methodologies. However, managed realignment can be cost-effective, partly as a result of lower maintenance costs and particularly when habitat values are added to the equation. Value may be further enhanced through the role of managed realignment schemes in reducing flood defence costs elsewhere in an estuary or along a coast. The specific details of each case especially the geography of the sites and coastal/fluvial processes, the pattern and state of existing defences, and the human activities protected are of crucial importance to the determination of cost-effectiveness of different options for flood defence.

Esteves (2014) stated that the Environment Agency typically worked to a guideline cost of £50,000 per hectare for managed realignment schemes but that many have questioned this cost. A review of the cost of managed realignment over the past 20 years in the UK has shown that costs per hectare have varied widely depending on factors such as the size of the scheme, the promoter and the date of implementation (Rowlands 2011). Costs are likely to be much higher where remediation of contaminated land is required (Latham et al. 2013, Park 2013). A review of the likely costs of managed realignment in Wales shows costs ranging from £100,000 to £675,000 per hectare (Park 2013). The price of land is also a factor; for example, high prices in areas such as the Thames Estuary may negatively affect scheme viability.

In the right setting, managed realignment can be a cost-effective coastal management measure for FCERM (as demonstrated, for example, at Medmerry) and this is the main reason why managed realignment policies are often adopted by SMPs (Esteves 2014).

5.4.5 Key case studies

Managed realignment

A significant number of managed realignment schemes have been implemented in the UK (Table 5.5). These are listed in OMReg, from which most information is taken, supplemented in some cases by the author's personal knowledge. Between them, these schemes illustrate the range of purposes for and locations at which managed realignment has been undertaken. It is important to note, however, that most schemes have multiple benefits and provide both flood risk management and biodiversity gain.

Of the 42 known British schemes:

- 26 include 'habitat creation' or 'biodiversity gain' among their objectives
- 18 include 'flood protection' or 'flood risk management' among their objectives
- 14 include 'compensation' or 'compensatory habitat' among their objectives

- 11 include 'reducing flood management costs' among their objectives
- 11 include 'creating more natural shoreline' among their objectives
- 6 include 'demonstration' or 'pilot' among their objectives
- 3 include 'accommodation space' or 'flood storage' among their objectives
- 2 give 'unknown' as their objective
- 1 includes 'recreational facility' among its objectives

One British managed realignment scheme not included in the OMReg database is Fingeringhoe in Essex. This was implemented in 2015 and comprises 22ha of saltmarsh, transitional habitat and mudflat. The purpose was habitat compensation, habitat creation and creation of a more natural shoreline.

Table 5.6 lists 48 managed realignment schemes undertaken elsewhere in western Europe. Data are again sourced from OMReg. Of these:

- 23 include 'habitat creation' or 'biodiversity gain' among their objectives
- 22 include 'compensation' or 'compensatory habitat' among their objectives
- 16 include 'creating more natural shoreline' among their objectives
- 3 include 'reducing flood management costs' among their objectives
- 3 include 'unknown' among their objectives
- 2 include 'flood protection' or 'flood risk management' among their objectives
- 2 include 'climate change' among their objectives
- 2 include 'water quality' among their objectives
- 1 includes 'mitigation' among its objectives
- 1 includes 'demonstration' or 'pilot' among its objectives

It is noticeable that objectives relating to flood risk management and reducing flood management costs are represented in fewer overseas schemes (10%) than UK schemes (69%). Objectives relating to habitat creation or compensation are important both overseas (94% of schemes) and in UK (95% of schemes). However, there are fewer multiple objectives recorded for overseas projects, which may reflect differences in the way data were collected and in the level of knowledge that the database compilers had of the schemes.

Rupp-Armstrong and Nicholls (2007) compared the application of managed realignment in England and Germany. They noted that managed realignment in England, and to some extent in Germany, seemed to be driven by longer term factors such as the desire to create more sustainable flood defence and to provide new intertidal habitats, although these are often combined with the more immediate need to upgrade defences. On Germany's well-defended North Sea coast, managed realignment has only been undertaken for specific compensation reasons, though broader conservation concerns might become an important future driver for managed realignment in summer polders.

From the personal knowledge of both the author and the reviewer, most UK managed realignment schemes are primarily driven by habitat creation or compensation with only a minority providing flood risk management benefits as a primary (rather than a secondary) objective.

Regulated tidal exchange

Table 5.7 lists 22 known regulated tidal exchange schemes in Great Britain, for which data have been sourced from OMReg. Of these:

- all include 'habitat creation', 'habitat enhancement', 'habitat protection' or 'biodiversity gain' among their objectives
- 4 include 'demonstration' or 'pilot' among their objectives
- 3 include 'flood protection' or 'flood risk management' among their objectives
- 3 include 'compensation' or 'compensatory habitat' among their objectives
- 3 include 'recreational facility' among their objectives
- 2 include 'mitigation' among their objectives
- 2 include 'reducing flood management costs' among their objectives
- 2 include 'climate change (sea level rise)' among their objectives
- 1 includes 'flood storage' among its objectives

Compared with British managed realignment schemes, a higher proportion (100%) of regulated tidal exchange schemes have objectives relating to habitat creation or compensation and a lower proportion (22%) have objectives relating to flood risk management and reducing flood management costs.

In practice, the role of regulated tidal exchange in flood risk management is limited as it requires the existing line of defence to be held. Possible FCERM roles are that regulated tidal exchange may contribute to:

- reducing estuary water levels (though the regulated nature of the tidal exchange is likely to limit this in an extreme event)
- accreting sediment as a precursor to full managed realignment

5.4.6 Funding

Where undertaken by public authorities, managed realignment is usually funded through FCERM grant-in-aid, either as a FCERM measure itself and/or as compensatory habitat to offset the impact of other FCERM schemes. This is the most common source of capital funding for managed realignment schemes.

Private organisations such as port developers also fund managed realignment to deliver compensatory intertidal habitat, for example, the Welwick (Yorkshire) and Chowderness (Humber) schemes funded by Associated British Ports.

Another funding mechanism is provided by the Higher Tier of Defra's Countryside Stewardship. This can pay feasibility, capital and management costs in relation to eligible habitat creation, including:

- CT7: Creation of intertidal and saline habitat on intensive grassland
- CT6: Coastal vegetation management supplement
- CT5: Creation of intertidal and saline habitat by non-intervention
- CT4: Creation of intertidal and saline habitat on arable land

• CT3: Management of coastal saltmarsh

Although public authorities are not eligible to claim Countryside Stewardship, tenant farmers and NGOs such as Wildlife Trusts are able to do so for land that is leased to them. For example, following creation of 30ha of priority habitats including saltmarsh, the Environment Agency leased Rye Harbour Farm to Sussex Wildlife Trust which draws on £50,000 annual stewardship payments, meeting 38% of the annual running costs. The Environment Agency has also facilitated existing landowners in claiming stewardship payments to fund feasibility studies into managed realignment on their land, for example, at Lower Clyst Estuary in Devon.

EU funding such as LIFE-Nature is another potential source of funds for managed realignment, where habitat creation and/or restoration is a primary objective.

5.4.7 Design, management and maintenance

Managed realignment is generally a significant engineering project requiring feasibility studies, an EIA, outline and detailed design. Capital costs of implementing schemes typically range from £17,000 per hectare to £60,000 per hectare (ABPMer 2015), for example:

- Wallasea Wild Coast Project: £22,500 per hectare (Environment Agency 2008b)
- Alkborough: £25,000 per hectare (Environment Agency 2009)
- Steart Coastal Management project: £62,500 per hectare (based on total cost of £20.2 million (Environment Agency 2014, personal communication) and 323ha of intertidal habitat)

An example of criteria for site selection is given in relation to the Ribble Estuary in Environment Agency (2004). Potential factors in site selection include:

- potentially contaminated land/landfill
- infrastructure including commercial/residential properties and roads
- · potential for adverse physical process implications
- new sea defences are required
- Grade 1, 2 or 3 agricultural land
- commercial fisheries
- SACs and candidate SACs (cSACs)
- Special Protection Areas (SPAs) and proposed SPAs (pSPAs)
- Ramsar sites
- SSSIs
- county Wildlife Sites
- Landscape Environmentally Sensitive Areas
- Green Belt
- Country Parks adjoining sites
- Scheduled Monuments

Table 5.5

Managed realignment schemes in Great Britain

Scheme name	Location	Total	Year	Habitat created	Primary purpose of realignment
		area	opened		
Alkborough Managed	Humber	440ha	2006	Saltmarsh, mudflat, transitional	Accommodation space in estuary
Realignment				grassland, reedbeds, lagoon	Also: improve flood protection,
					compensation, habitat creation
Abbotts Hall Farm	Essex	80ha	2002	Saltmarsh, transitional	Biodiversity gain
				grassland, mudflat	Also: improve flood protection, habitat
					creation, create more natural shoreline,
		1001			reduce flood protection costs
Allfleets Marsh	Essex	133ha	2006	Mudflat, saltmarsh, lagoon,	Compensation, create more natural
(Wallasea Island)				grassland	shoreline, improve flood protection
Alnmouth 1	Northumberland	8ha	2006	Saltmarsh	Reduce flood protection costs
Alnmouth 2	Northumberland	20ha	2008	Saltmarsh, grassland	Habitat creation
Annery Kiln	Devon	3.8ha	2000	Saltmarsh	Habitat creation, create more natural
					shoreline
Barking Creek	London	0.039ha	2006	Transitional grassland	Habitat creation, improve flood
					protection
Barking Creek	London	1ha	2006	Saltmarsh, mudflat	Habitat creation, demonstration/ pilot,
(Barking Barrier)					flood storage provision
Bleadon Levels	Somerset	13ha	2001	Saltmarsh	Habitat creation, improve flood
					protection
Brancaster West Marsh	Norfolk	7.5ha	2002	Saltmarsh	Reduce flood protection costs, Habitat
					creation, improve flood protection
Brandy Hole	Essex	12ha	2002	Saltmarsh, transitional	Compensation, habitat creation
				grassland	
Chowder Ness	Lincolnshire (Humber)	15ha	2006	Mudflat, saltmarsh, transitional	Compensation, habitat creation
				grassland, terrestrial grassland	
Cobnor Point	Sussex	6.5ha	2013	Mudflat, saltmarsh, terrestrial	Compensation
				grassland	
Cone Pill	Gloucestershire	50ha	2001	Terrestrial grassland	Reduce flood protection costs
	(Severn)				
Devereaux Farm	Essex (Hamford	15ha	2010	Saltmarsh, lagoon	Habitat creation, compensation
(Phase 1)	Water)				

Scheme name	Location	Total	Year	Habitat created	Primary purpose of realignment
		area	opened		
Freiston	Lincolnshire (The Wash)	66ha	2002	Saltmarsh	Create more natural shoreline, reduce flood protection costs, habitat creation
Halvergate	Norfolk (Yare)	0.5ha	2005	Lagoon, saltmarsh, reedbeds	Unknown, improve flood protection
Havergate Island	Suffolk	8.1ha	2000	Mudflat, saltmarsh	Habitat creation, improve flood protection
Hesketh Outmarsh West	Lancashire	180ha	2008	170ha saltmarsh, mudflat, transitional grassland, lagoon	Habitat creation, improve flood protection, compensatory habitat
Kennet Pans	Fife (Firth of Forth)	8.2ha	2007	Mudflat, saltmarsh, terrestrial grassland	Compensation
Lantern Marsh	Suffolk	29ha	1999	Saltmarsh, mudflat	Reduce flood protection costs, create more natural shoreline, improve flood protection
Medmerry Managed Realignment	West Sussex	450ha	2013	183ha mudflat, lagoon, saltmarsh Also terrestrial grassland, transitional grassland	Compensation, flood risk management, reduce flood protection costs, create more natural shoreline, recreational facility
Millennium Terraces	London (Thames)	0.5ha	1998	Mudflat, saltmarsh, reedbeds	Habitat creation, demonstration/ pilot
Montrose Basin	Angus	0.3ha	1997	Saltmarsh	Improve flood protection, Reduce flood protection costs
Nigg Bay (Meddat Marsh)	Highland (Cromarty Firth)	25ha	2003	Saltmarsh, transitional grassland, mudflat	Habitat creation, demonstration/ pilot, create more natural shoreline
Northey Island	Essex (Blackwater)	0.8ha	1991	Saltmarsh, lagoon	Habitat creation, demonstration/ pilot
Orplands	Essex (Blackwater)	38ha	1995	Saltmarsh, mudflat	Reduce flood protection costs, improve flood protection, create more natural shoreline
Paull Holme Strays	Yorkshire (Humber)	80ha	2003	Saltmarsh, mudflat, transitional grassland	Improve flood protection, compensation, habitat creation
Pawlett Hams	Somerset (Parret)	4.8ha	1994	Mudflat	Reduce flood protection costs
Pillmouth (Phase 1 and 2)	Devon (Torridge)	12.9ha	2001	Saltmarsh	Habitat creation, create more natural shoreline
Stanford Wharf	Essex (Thames)	27ha	2010	Mudflat	Compensation
Steart Coastal	Somerset (Parret)	400ha	2014	183 of intertidal saltmarsh	Compensatory habitat, flood storage
Management Scheme				40ha of intertidal mudflat	provision, habitat creation

Scheme name	Location	Total	Year	Habitat created	Primary purpose of realignment
		area	opened		
				69ha of transitional brackish	
				habitat	
				32ha of brackish and saline	
				lagoons	
Thorness Bay	Isle of Wight (Solent)	7ha	2004	Saltmarsh, transitional	Habitat creation
				grassland, lagoon	
Thornham Point	West Sussex	6.9ha	1997	Saltmarsh, transitional	Habitat creation, reduce flood protection
	(Chichester Harbour)			grassland	costs
Titchwell Marsh	Norfolk	11ha	2011	Mudflat, saltmarsh	Flood risk management
Tollesbury	Essex (Blackwater)	21ha	1995	Saltmarsh, mudflat	Improve flood protection,
					demonstration/pilot, habitat creation
Trimley Marsh	Suffolk (Orwell)	16.5ha	2000	Mudflat, saltmarsh	Compensation, habitat creation
Tutshill	Somerset	2ha	2011	Saltmarsh	Unknown
Wallasea Island	Essex	165ha	2015	Mudflat, saltmarsh, transitional	Flood risk management, create more
(Phase 1 – Jubilee Marsh)				grassland	natural shoreline, habitat creation,
					demonstration/pilot
Watertown Farm	Devon	1.5ha	2000	Saltmarsh	Habitat creation, create more natural
					shoreline
Welwick	Yorkshire (Humber)	54ha	2006	Mudflat, saltmarsh, transitional	Compensation
				grassland, lagoon	
Ynys-hir	Ceredigion	6ha	2010	Saltmarsh	Flood risk management and biodiversity
					gain

Table 5.6

Managed realignment schemes in western Europe

Scheme	Location	Total	Year	Habitat created	Primary purpose of realignment
		area	opened		
Heusden	Zeeschelde, Belgium	13ha	2006	Saltmarsh, lagoon	Compensation, habitat creation,
					improve flood protection
Ketenisseschor	Zeeschelde, Belgium	36ha	2002	Saltmarsh, mudflat	Compensation
Lillo-Potpolder	Zeeschelde, Belgium	8ha	2012	No information	Unknown
Paardeschor	Zeeschelde, Belgium	12ha	2004	Saltmarsh, mudflat	Compensation
Paddebeek	Zeeschelde, Belgium	1.6ha	2003	Saltmarsh, mudflat	Compensation
Yzer Mouth	Yzer, Belgium	50ha	2001	No information	Compensation
Gedal Strandenge	Limfjord, Denmark	140ha	1992	No information	Unknown
Viggelso	Odense Fjord, Denmark	66ha	1993	No information	Unknown
Aber de Crozon	Crozon, France	90ha	1980	No information	Habitat creation
Anklamer Stadtbruch	Oderhaff, Germany	1,750ha	2004	Transitional grassland, lagoon	Reduce flood protection costs, habitat creation
Billwerder Insel	Elbe Estuary, Germany	20ha	2008	Reedbeds, mudflat	Compensation, habitat creation
Dorumer Sommerpolder	East Friesland, Germany	4ha	2001	Saltmarsh	Compensation
Hahnofer Sand	Elbe, Germany	104ha	2002	Mudflat, lagoon, saltmarsh	Compensation
Hauener Hooge	Ley Bay, Germany	80ha	1994	Saltmarsh	Compensation, habitat creation, create more natural shorelines
Karrendorfer Wiesen	Griefswald Boddewn, Germany	350ha	1993	Saltmarsh, transitional grassland, terrestrial grassland	Habitat creation, climate change (greenhouse gas) mitigation, reduce flood protection costs
Kleinensieler Plate	Weser, Germany	58ha	2000	Transitional grassland, lagoon, reedbeds	Compensation
Kleines Noor	Flensburg Fjord, Germany	14ha	2002	Lagoon	Habitat creation, improve water quality
Kreetsand	Elbe, Germany	26ha	1999	Transitional grassland, reedbeds, mudflat	Compensation, habitat creation, create more natural shoreline
Scheme	Location	Total	Year	Habitat created	Primary purpose of realignment
--------------------------------------	--------------------------	-------	--------	----------------------------------	-----------------------------------
		area	opened		
Langeooger Sommerpolder	Island of Langeoog,	215ha	2004	Saltmarsh	Compensation, habitat creation,
	Germany				create more natural shoreline
Lütetsburger	East Friesland, Germany	15ha	1982	Saltmarsh	Compensation, habitat creation,
Sommerpolder					create more natural shoreline
Pepelower/Tessmansdorfer	Salzhaff, Germany	120ha	2002	Reedbeds, saltmarsh	Habitat enhancement
Wiesen					
Polder Freetz	Island of Rügen, Germany	180ha	2002	Saltmarsh	Compensation, create more natural
<u> </u>	<u> </u>	0.01	1000		shoreline
Polder Friedrichshagen (Ziesetal)	Greifswald Bodden,	90ha	1999	Saltmarsh	Habitat creation, climate change
	Cermany				costs
Polder Neuensien	Island of Rügen, Germany	40ha	2002	Saltmarsh	Compensation, create more natural
(Südwestteil)					shoreline
Polder Roggow	Salzhaff, Germany	40ha	2002	Reedbeds, saltmarsh	Habitat creation
Polder Wehrland	Peenestrom, Germany	113ha	2004	Lagoon, saltmarsh, transitional	Compensation
				grassland	
Riepenburg	Elbe, Germany	1ha	1995	Reedbeds, mudflat, transitional	Compensation, habitat creation,
				grassland	create more natural shoreline
Rönnebecker Sand	Weser, Germany	34ha	2002	Lagoon, reedbeds	Compensation
Sommerpolder Wurster	East Friesland, Germany	145ha	2007	Saltmarsh, transitional	Compensation, create more natural
Küste				grassland, mudflat	shoreline
Spadenländer Spitze	Elbe, Germany	7.5ha	2000	Transitional grassland, mudflat,	Compensation, habitat creation,
				reedbeds	create more natural shoreline
Strandseenlandschaft	Kiel Bay, Germany	40ha	1989	Lagoon, saltmarsh, transitional	Compensation
Schmoel				grassland	
legeler Plate Polder	Weser, Germany	150ha	1997	Mudflat, reedbeds	Compensation, create more natural
	Deensetrem, Correction	CCha	2004	Largen eeltmareh trensitional	
Telipoider Waschow	Peenestrom, Germany	66na	2004	grassland	Compensation
Vorder/Hinterwerder Polder	Weser, Germany	30ha	1997	Reedbeds, lagoon, terrestrial	Compensation, habitat creation
				grassland	
Wrauster Bogen	Elbe, Germany	2.2ha	1991	Reedbeds, transitional	Compensation, habitat creation,
				grassland, mudflat	create more natural shoreline

Scheme	Location	Total	Year	Habitat created	Primary purpose of realignment
		area	opened		
Bildtpollen	Mainland coast of	60ha	2009	No information	Habitat creation, demonstration/pilot
	Friesland, Netherlands				
Bunkervallei, de Slufter	Island of Texel, Netherlands	3ha	2002	Dunes	Habitat creation
de Kerf	Mainland coast of North Holland, Netherlands	30ha	1997	Dunes, saltmarsh, lagoon	Create more natural shoreline, compensation, demonstration/pilot
Groene Hoek, de Slufter	Island of Texel, Netherlands	13ha	2002	Dunes	Habitat creation
Groene Strand	Island of Terschelling, Netherlands	23ha	1996	Transitional grassland, saltmarsh	Habitat creation, create more natural shoreline
Holwerder Zomerpolder	Mainland coast of Friesland, Netherlands	28ha	1989	Saltmarsh	Create more natural shoreline
Klein Profijt	Oude Maas, Netherlands	6ha	2005	Mudflat, saltmarsh	Improve water quality, habitat creation
Kroon's Polders	Island of Vlieland, Netherlands	85ha	1996	Saltmarsh, mudflat, lagoon, dunes	Habitat creation, create more natural shoreline
Noard Fryslân Bûtendyks	Mainland coast of	135ha	2001	Saltmarsh	Habitat creation, create more
	Friesland, Netherlands				natural shoreline, improve flood
					protection
Sophiapolder	Noord, Netherlands	77ha	2012	No information	Habitat creation
Tiengemelen	Haringvliet, Netherlands	450ha	2007	Saltmarsh, reedbeds, lagoon, transitional grassland, terrestrial grassland	Habitat creation
Ría de la Rabia	Oyambre, Spain	47ha	2009	Saltmarsh	Create more natural shoreline, habitat enhancement
Vega de Jaitzubia	Bidasoa/Jaitzubia, Spain	23ha	2004	Saltmarsh	Habitat creation

Table 5.7

7 Regulated tidal exchange schemes in Great Britain

Scheme	Location	Total	Year	Habitat created	Primary purpose of realignment
		area	opened		
Black Devon Wetlands	Clackmannanshire	28ha	2000	Lagoon, transitional grassland	Habitat creation, demonstration/pilot
Black Hole Marsh	Axe, Devon	6ha	2009	Lagoon, transitional grassland	Habitat creation, flood storage provision, recreational facility
Bowers Marsh	East Haven Creek, Essex	10ha	Not known	Lagoon, saltmarsh	Habitat enhancement
Chalkdock Marsh	Chichester Harbour, West Sussex	3.3ha	2000	Saltmarsh	Habitat creation
Clapper Marshes	Camel, Cornwall	10ha	2011	Saltmarsh	Habitat creation
Glasson	Conder, Lancashire	6.4ha	2005	Lagoon, transitional grassland, saltmarsh	Reduce flood protection costs
Goosemore	Clyst, Devon	6.3ha	2004	Saltmarsh, mudflat, lagoon	Habitat creation, demonstration/pilot, habitat enhancement
Goswick Farm (Beal)	South Low River, Northumberland	4.5ha	2010	Saltmarsh, lagoon	Habitat creation
Horsey Island	Hamford Water, Essex	1.2ha	1995	Saltmarsh	Habitat creation
Lepe	Dark Water, Hampshire	4ha	2006	Saltmarsh	Habitat creation, improve flood protection, compensation
Lymington Estuary	Hampshire	21ha	2009	Reedbeds, mudflat	Compensation, climate change (sea level rise) mitigation, habitat enhancement
Ryan's Field	Hayle, Cornwall	6.2ha	1995	Saltmarsh, lagoon, rocky shore	Habitat creation
Rye Harbour Farm	East Sussex	17ha	2011	Saltmarsh and lagoon	Biodiversity gain
Saltram	Plym, Devon	4.2ha	1995	Saltmarsh	Habitat creation
Seal Sands	Tees, Durham	9ha	1993	Saltmarsh, mudflat	Habitat creation, compensation
Skinflats	Firth of Forth, Falkirk	11ha	2009	Saltmarsh, lagoon, terrestrial grassland	Habitat creation, demonstration/pilot, recreational facility
South Efford Marsh	Avon, Devon	17ha	2011	Saltmarsh, transitional grassland	Habitat creation
Treraven Meadows	Camel, Cornwall	14ha	2007	Saltmarsh, transitional grassland	Habitat creation
Vange Marsh	Thames, Essex	1ha	2006	Lagoon, rocky shore, transitional grassland, saltmarsh	Habitat creation
Walborough	Axe, Somerset	4.5ha	2004	Saltmarsh, mudflat, lagoon	Improve flood protection, reduce flood protection costs, habitat creation

Scheme	Location	Total	Year	Habitat created	Primary purpose of realignment
		area	opened		
Warkworth	Coquet, Northumberland	0.4ha	2009	Saltmarsh	Habitat creation
West Wittering	Chichester Harbour, West Sussex	6ha	2013	Lagoon, saltmarsh	Flood risk management, climate change (sea level rise) mitigation, habitat protection, improve flood protection, recreational facility

- listed buildings
- public rights of way
- non-scheduled known archaeological sites
- historic landscape features

CIRIA (2004) provides a manual of design issues relating to managed realignment. Central to the design of any managed realignment project are its interactions with physical processes and morphology, in terms both of their impact on a scheme and a scheme's impact on them. A combination of desk-based reviews, field-based measurements and numerical model-based assessments are needed to understand the implications for design. The most important issues to be considered are:

- tidal levels and range determines the flooding regime of a site and likely habitat formation
- tidal prism and discharges –affects the impact of managed realignment on estuary dynamics
- tidal current velocities –affects sediment deposition and erosion
- tidal asymmetry, that is, whether the estuary is ebb or flood dominated and the implications for sediment availability
- wave climate may affect sediment deposition and erosion and the likely habitat formation
- sediment dynamics and morphological response depends on the above factors

Design considerations for a realignment scheme include:

- definition of site boundaries (extent of realignment)
- need for and specification of realigned defence
- treatment of new intertidal area creek systems, existing vegetation, buildings, structures and services
- presence of any waste or contaminated land that may have to be removed or contained
- design of breaches, especially the size of breach and the channels required to convey water into and out of the realigned area
- establishment of intertidal vegetation by planting or natural colonisation
- flood risk assessment and effects on existing land drainage, including any requirement for new culverts to deal with freshwater drainage
- decommissioning of existing defences
- health and safety issues

Unless armoured, breaches will tend to rapidly reach equilibrium size as a result of erosion by water passing through them. Allowing natural scour may be undesirable owing to the volumes of sediment released. Breach size can be optimised during design by hydraulic modelling.

Maintenance and management is also likely to be required, such as controlled grazing to maximise the botanical and ornithological diversity of saltmarsh. Because of its

intrinsic wildlife value, management is frequently of interest to conservation organisations such as the RSPB, the Wildfowl & Wetlands Trust and county Wildlife Trusts.

Monitoring is an important aspect of a managed realignment project. Its aim is to assess the project's impacts and to determine whether the design is operating as intended. Important issues for monitoring include:

- geomorphological change such as water levels, currents and sediment accretion rates
- water quality, especially sediment loading
- change in flora, including the establishment of saltmarsh vegetation (if that is the objective)
- change in fauna, such as colonisation by intertidal invertebrates and the use of the site by birds for wintering, feeding and breeding
- change to the landscape
- changes in public use, such as increases or decreases in recreational visits
- condition of defences that may be affected, including realigned defences and downstream defences

5.4.8 Other requirements

Obtaining consents such as planning permission and marine licenses for managed realignment can be complex. Often many stakeholders are involved and schemes can be controversial with local people and/or elected representatives. Many schemes are located within or adjacent to environmentally sensitive areas such as SSSIs, SPAs for birds and SACs. Existing freshwater habitat may be lost to saline inundation and existing intertidal habitat may be modified by the creation of new creeks. Groundwater may be affected by salinity. Known or potential archaeological sites may be affected by saline inundation. Footpaths may be severed and require re-routing. The landscape will be changed and views may be gained or lost as a result of defences being realigned. Numerical modelling will be required to understand the effects of managed realignment on coastal processes and estuarine water levels.

The Essex Wildlife Trust (2005) reported a number of difficulties in the planning process for managed realignment at Abbotts Hall Farm – one of the first managed realignment schemes to be implemented. Lessons learned included the following.

- Essex Wildlife Trust was not well versed in the role of 'developer' and so approached planning too passively.
- The local council lacked experience in managed realignment and did not have the expertise to process the application confidently. Independent advice to the council was needed.
- Once the planning application was made, Essex Wildlife Trust could not talk to certain councillors about the project because it could be construed as lobbying to sway the planning process. These conversations needed to have taken place in advance.

Subsequent experience of many managed realignment schemes shows that the best route to successfully negotiating consents entails:

• early and thorough stakeholder participation during scheme development

 thorough preparation of documentation supporting consent applications, such as plans, outline designs, flood risk assessments, Water Framework Directive assessments, Habitats Regulations assessments and environmental assessments

An analysis found that SMPs had proposed setting back nearly 10% of the coastline by 2030, rising to nearly 15% by 2060. Achieving this goal would mean realigning around 30km of coastline every year to 2030. It would create around 6,200ha of coastal habitat by 2030 and 11,500ha by 2060 (Adaptation Sub-Committee 2013).

5.4.9 Research gaps

Managed realignment has become a well-established technique in the UK and overseas, with many examples of implemented schemes, some on a large scale. However, the principal driver for most implemented schemes has been intertidal habitat creation. In only a small number of cases are data available on FCERM benefits from managed realignment.

Demonstrating the FCERM benefits of managed realignment is complex owing to the many site-specific variables and differences in appraisal methodologies. In general, managed realignment appears most likely to be cost-effective as an FCERM measure:

- when account is taken of reduced long-term defence maintenance requirements
- in cases where provision of accommodation space within estuaries reduces pressure on defences elsewhere
- when account is taken of the ecosystem services value and compensatory value of intertidal habitat created

There is a need for more information on how to calculate FCERM and ecosystem service benefits from managed realignment. This might increase the opportunities to promote managed realignment as a NFM technique for a wider range of purposes in addition to providing compensatory habitat.

Important research gaps include:

- improved techniques for predicting habitat formation
- confidence in the sustainability of saltmarsh over the life of FCERM schemes and approaches to adapting FCERM design
- evidence for the role of intertidal habitat in improving water quality
- improved techniques and data availability for ecosystem services valuation, including techniques for incorporation into economic appraisals for schemes

5.5 Beach recharge/nourishment and sandscaping

5.5.1 Understanding the science

A healthy beach is one of the most effective forms of sea defence because, provided it is not constrained by space or material supply, it dissipates wave energy and has the ability to adapt its shape naturally to changing wave and tidal conditions (CIRIA 2010).

Many beaches have reduced in volume over time, often because of human activity such as removing beach material, levelling beach crests, protecting cliffs from erosion (hence reducing the supply of material) and installing structures such as groynes. Reductions in beach levels and widths reduce their effectiveness in preventing erosion or flooding, and adversely affect their contribution to landscape and habitat value.

Beach recharge or nourishment is the process of adding material to the shoreline, where it will be incorporated into a beach by natural processes and increase the effective standard of flood protection. It may also increase landscape and habitat value, providing additional benefits and cost justification.

There are 5 main potential sources of beach recharge material (CIRIA 2010):

- existing licensed offshore aggregate dredging areas
- offshore seabed deposits for which a specific dredging licence needs to be obtained
- navigation dredging operations, for example, at ports and harbours
- natural inland sand and gravel deposits and quarried rock
- secondary aggregates, for example, byproducts of industrial processes or materials from demolition or excavation

In some cases, source material for beach recharge may be a combination of these sources. For example, recharge of the lower shore at the Pevensey Bay frontage in East Sussex has been carried out since 2007 by dredging material previously deposited offshore following maintenance dredging of Sovereign Harbour (Pevensey Coastal Defence 2015). This material originally formed part of the same sediment cell, and could have reached Pevensey Bay naturally had it not been entrained in the entrance to the artificial Sovereign Harbour. This recharge is a distinct activity to the shingle recharge and bypassing also performed on the frontage.

Beach recharge can be carried out with shingle and/or sand. Many beaches in southeast England comprise a mixture of sand and shingle, whereas in south-west England beaches tend to be either sand or shingle. Recharge of mudflats can also be undertaken.

Shingle

'Shingle' describes beaches composed mainly of rounded pebbles, larger than sand (>2mm) but smaller than boulders (<200mm). Particle size distributions usually refer to more precise terms 'gravel' (>2mm and <60mm) and 'cobbles' (>60mm) (CIRIA 2010). Shingle is generally associated with high energy coasts where the waves are strong enough to move the material along and across the beach, sometimes above the limits of the tides. Shingle beaches are a widespread, naturally occurring coastal defence, particularly in south-east England and East Anglia. Mobile shingle beaches are good at absorbing wave energy.

There has been a tendency in the past for coastal engineers to regard some shingle beaches as if they were solid flood defence structures. In a number of locations (for example, formerly at Cley/Salthouse in Norfolk), shingle beaches have been profiled into high, narrow ridges to increase the standard of flood protection they provide. However, this increases their vulnerability to breaching during large surge events and leads to long-term loss of material, which is ultimately unsustainable. In order to work with natural processes, shingle beaches need to be recognised as energy absorbers and allowed to roll landwards in response to natural forcing factors. Shingle barriers will often migrate landwards in response to sea level rise. Their overtopping is part of the process that allows the barriers to move while maintaining their volume through a process of rollover. Allowing this migration is essential to their long-term stability. As part of the rollover and migration process, areas of vegetated shingle may become mobilised and the vegetation will disappear temporarily until it become re-established when the beach stabilises. Where sediment supply is sufficient, seaward progradation of shingle barriers may also occur. It is this process that has formed the many spits and nesses around the British coast. The implication of WWNP is that either landward or seaward movement of beaches may occur, or both at different times, and should be accepted.

Vegetated shingle is an important nature conservation feature and one which is nationally rare. The extent of coastal vegetated shingle in England has been estimated as 4,495ha (Houston et al. 2008), of which 95% is within an SSSI designation. JNCC gives areas of 3,596ha in England, 5,800ha in Wales and 2,045ha in Scotland (JNCC 2016). Vegetated shingle has 2 distinct vegetation types recognised at European level, that is:

- perennial vegetation of stony banks
- annual vegetation of drift lines

The overview of the management of coastal vegetated shingle in the UK by Randall (2004) outlined the key threats to coastal vegetated shingle and discussed measures to prevent damage or to offset unavoidable damage arising from coastal engineering works. Randall (2004) identified 2 main 'states' for shingle habitats and 3 further states. The 2 main states are termed:

- eroding
- accretional or semi-stable

The other states are:

- stable
- disturbed
- excavated

In general, the presence of vegetation on shingle is not considered to contribute significantly to its FCERM function, although it could play some role in attenuating waves during extreme events that overtop shingle beaches. Vegetated shingle is not considered separately in this report.

Sand

'Sand' describes sediments larger than silt (>0.06mm) and smaller than gravel (<2.0mm). Sandy beaches are naturally occurring defence structures that are widespread around the UK coast. Sandy beaches are important to FCERM both in their own right as an integral part of dune systems and in protecting the foundations of seawalls or other structures from being undermined.

Sandy beaches are often important for recreational amenity. While the habitat value of a sandy beach is generally less than that of mudflat or vegetated shingle, associated dune systems are often of very high nature conservation value.

A recent extension of the concept of beach recharge is the 'sand engine' or 'sand motor'. It originated in the Netherlands and is referred to as 'sandscaping' in the UK. This approach consists of large-scale beach nourishment that makes active use of

natural processes to feed and redistribute sediment to a coastline in order to meet FCERM objectives, while at the same time generating additional social, economic and environmental benefits (Royal HaskoningDHV 2015a).

The sandscaping approach has been pioneered by the Netherlands as part of the country's 'Building with Nature' innovation programme. This programme, which has been embraced by a significant consortium of government and other organisations, aims to integrate infrastructure, society and nature in new forms of engineering to achieve sustainability, particularly within the water environment. The most prominent case study of a fully implemented sand engine in the Netherlands is the Delfland Sand Engine. A pilot implemented along part of the Zuid-Holland coastline in 2011, the scheme involved recharging 20 million m³ of sand. It is reported Royal HaskoningDHV 2015a) to have generated benefits in terms of:

- recreation through the creation of new land with new opportunities
- efficiency through economies of scale (though it is reported that a factor in the cost of the scheme was the availability of the Dutch dredging fleet following a downturn in work in the Middle East)
- environmental enhancement in the nourished area and the sediment extraction area
- FCERM

According to Deltares (2013), the Delfland Sand Engine cost €70 million. The report by Deltares stated that mega-nourishment was not more cost-effective than regular small-scale nourishment in terms of its primary function (coastal management for flood protection). Indeed, the latter was quicker to yield a return on the investment (Deltares 2013). Mega-nourishment creates added value mainly for recreation and nature, and potentially also for drinking water supplies.

Royal HaskoningDHV (2015b) identified 12 sites in England which were considered to have high potential for sandscaping. The main benefits of a sandscaping scheme were identified as generally relating to:

- flood risk reduction and protection
- economic and tourism benefits
- amenity improvements
- habitat improvement
- place-making
- enhancing the identity of a location

The report did not fully consider the advantages and disadvantages of this approach, such as the potential negative effects on existing habitats, including those protected by designations, widescale changes to coastal processes and effects on water quality.

The cost-effectiveness of this approach in the UK, where FCERM funding differs from that in the Netherlands, has not been demonstrated. It should also be noted that the intention of sandscaping is to modify coastal processes and so it is debatable to what extent it represents WWNP. More objective analysis is needed for further consideration to be given to adopting sandscaping.

Mud

'Mud' refers to beaches that predominantly consist of silt (particles >0.002mm and <0.06 mm) and clay (particles <0.006mm).

Recharge of saltmarsh and mudflats with fine sediment may be carried out to improve their FCERM performance. More usually, however, it is aimed primarily at enhancing or restoring intertidal habitats and finding a beneficial use for dredged material derived from maintaining navigation channels.

The Environment Agency recently carried out a project to produce a toolkit for the beneficial use of dredged material in Essex and Suffolk (Environment Agency, undated b). This project examined opportunities to use dredging as mitigation to offset potential impacts to SSSIs within the proposed dredging area. For example, material from dredging campaigns in Harwich Haven has been used beneficially to recharge areas within Hamford Water and the Blackwater Estuary.

From the perspective of FCERM, there are 3 key issues to consider in relation to beach recharge (CIRIA 2010):

- beach stability (how long the recharge material will stay in place)
- sediment dynamics of placed material
- durability of recharge material on the beach

Beach recharge is often (though not always) carried out in combination with beach control structures to retain the new material in place. It is also sometimes undertaken in combination with beach profiling. In this case the recharge material is bulldozed into an unnaturally steep profile to increase the standard of flood protection.

Artificial profiling tends to lead to a long-term loss of beach material and an increase in the probability of breaching in a large storm event, and is not considered to represent WWNP. Factors such as an increasing frequency of severe wave conditions, storm surge intensity and limited sediment supply have often resulted in beach profiling becoming unsustainable and hence it is no longer recommended for many situations (CIRIA 2010). This is usually because there is insufficient volume of beach material within the system to sustain a constant beach alignment. Holding the line artificially results in the rollback of the system becoming out of phase with its natural evolution. Developing an adaptive management strategy for such a beach can be effective where the intention is to:

- control the rate of rollback
- revert to a more natural evolution
- protect natural or built assets
- make the system more sustainable

This requires an understanding of the beach cross-section, surge water levels, storm intensity and storm frequency.

5.5.2 Confidence in the science

Beach recharge is a well-understood technique that is widely practised. However, there remain areas of uncertainty as to how beach material will behave in practice on any given beach. Sources of uncertainty may include:

• suitability of recharge material

- how recharge material will behave under wave and tidal action
- beach stability and durability
- effects of sediment starvation
- effects of small-scale coastal processes (for example, small changes in nearshore wave climate can change the direction of longshore drift at some locations)
- effects of episodic events, sequences of events and storm surges
- areas of complex coastal processes such as those associated with entrances to tidal inlets or zones immediately downdrift of terminal structures
- effects of climate change

For example, recharging a beach with mixed sand and shingle will lead to an interim period during which material is worked and sorted by wave action, as a result of which the profile will alter. A commonly encountered problem with mixed sand–shingle recharge is 'cliffing', which is the formation of steep scarps where the mixed material is eroded as if it were a cliff rather than absorbing wave energy as a mobile beach. Such problems can be avoided by recharge using material of the right grading or addressed by removing/redistributing sediments using a bulldozer (CIRIA 2010).

Continuing approaches to beach management that have been practised for many years (for example, recycling or reprofiling) often restricts the natural evolution of beach systems. However, changes to more sustainable management solutions (allowing the beach to behave more naturally) may be hampered by uncertainty. Beach recharge and other forms of beach management involve WWNP, and consequently there is often significant uncertainty as to the outcome.

Numerical models can provide some basis for decision-making, but are generally not sufficiently sensitive to take account of small-scale localised changes (Environment Agency 2013b). A practical way to break into this 'cycle of uncertainty' is through adaptive management. This refers to making incremental changes and monitoring the results so as to improve understanding.

The context of adaptive management as applied to beach recharge enables complex coastal processes to be assessed by monitoring changes. Small adjustments and reactive responses to changes can be made to optimise natural processes. Good documentation of both actions and responses is required. The basic premise of WWNP is crucial to the success of this approach. The main action required to is to start the monitoring and adaptation process even if there is insufficient information to make an immediate diagnosis of the final solution (CIRIA 2010). Adaptive changes could include:

- minor beach realignment
- incremental addition of new beach material
- movement of beach material
- gradual changes to beach geometry

Attempts to reduce or increase transport rates and change beach plan alignment may also feature. As with other WWNP measures, beach management necessarily involves managing risk and uncertainty (CIRIA 2010).

5.5.3 Metrics of cost–benefit and multiple benefits

Beach recharge projects are generally justified on the basis of FCERM benefits using traditional benefit–cost analysis (CIRIA 2010). In common with other FCERM schemes, the economic benefit of the scheme is represented by the reduction in loss or damage (compared with the Do Nothing case) to:

- buildings and contents
- physical stock including agriculture
- costs of emergency service response
- loss of industrial production
- · disruption of communications and public utilities
- disruption of traffic
- loss of land and agricultural production
- recreational and societal impacts
- risk to life

Benefits to the natural environment such as improved or restored habitat (for example, vegetated shingle, sand dunes or mudflats) are generally accounted for within Defra's Outcome Measures. Benefits to recreation and amenity may be substantial in the case of beach recharge; the Multi-Coloured Manual (Penning-Rowsell et al. 2005) recommends that Value of Enjoyment (based on visitor questionnaires) be used in economic analysis.

Beach recharge has a number of implications for nature conservation. An increase in the area of stable beach substrate above mean high water spring level can encourage colonisation by vegetation. In the case of sandy substrate, this could include pioneer dune vegetation, while on shingle beaches it could comprise perennial vegetation of stony banks and annual vegetation of drift lines. An increase in beach elevation can also provide additional nesting habitat for, and increased nesting success by, shore-nesting birds such as little terns – also observed at Happisburgh to Winterton. All of these represent potentially significant nature conservation gains for protected and/or Priority habitats and species.

However, the placement of beach recharge material and associated beach profiling can be detrimental to nature conservation, for example, by disturbing or destroying beach vegetation. The impact depends on the volume, location and frequency of recharge. The more frequently beach recharge is undertaken, the greater the likely impact on the ecology of the beach as there is a shorter period available for the recovery of vegetation between recharge events.

Sandscaping is an innovative approach, which can entail greater costs than traditional beach recharge and potentially have wider ranging, albeit relatively diffuse, benefits and impacts. The north Norfolk sandscaping feasibility study (Royal HaskoningDHV 2015a) noted that, given the significant investment involved and the inherent complexities in developing a more integrated, broader scale approach, the opportunities that can be delivered by sandscaping can be lost within a more traditional project appraisal driven by short-term or immediate need. According to the report:

'The Sandscaping approach to coastal management can best be described as a large-scale beach nourishment that aims to maximise the beneficial role of natural processes. What distinguishes it from a more conventional linear nourishment of the shoreline (whether through annual nourishment or larger

scale linear nourishment), is that it is a more radical, shoreline-changing approach, designed to alter the geomorphology in such a way that the subsequent interaction with the natural processes enhances the protection it provides and the benefits that it generates' (Royal HaskoningDHV 2015a, p. 4).

Some of the main potential benefits of sandscaping are (Royal HaskoningDHV 2015a):

- protection of existing assets and functions
- improved efficiency of FCERM over a larger management area
- creation of width within the natural coastal system that builds in resilience to climate change
- creation of a blank canvas for new economic, social and environmental opportunities and development

Royal HaskoningDHV (2015a) noted the following benefits of sandscaping as applied in the pilot Delfland Sand Engine project in the Netherlands in 2011.

- The coastline had previously required nourishment every 5 years to maintain beach levels. The sand engine was designed to reduce that requirement to once every 20 years.
- Significant different recreation and tourism opportunities were created, broadening the attraction of the coast. Since the public has been able to access the sand engine, it has become very popular for kite surfing.
- The project provided more effective delivery of benefits, particularly in terms of the substantial reduction in costs of the approach compared with traditional beach nourishment schemes. This is a result of multiple factors, including economies of scale effects and the effective life of the scheme.
- Habitat disturbance has been reduced due to less frequent nourishments.
- New habitats have been created in both the borrow and the nourishment areas though this could also represent disturbance and damage to existing habitats.

The Delfland scheme saw the placement of 21.5 million m³ of sand in a hook shape, designed in such a way that processes will transport the sediment downdrift, building up the beaches and forming a coastal lagoon feature in the centre of the newly built-out area.

It is reported that a factor in the implementation of a sandscaping approach was the availability of the Dutch dredging fleet following a downturn in work in the Middle East (Pontee, personal communication). If this is the case it may have implications for the practicability of the approach elsewhere.

The economic benefits to tourism over 50 years of sandscaping between Mundesley and Walcott in Norfolk could amount to a present value of £339 million. The 30ha of new intertidal habitat that would be created was valued at £1.5 million (Royal HaskoningDHV 2015a). These figures would need to be subject to robust appraisal.

While sandscaping provides sediment that is worked on by natural processes, it might be questioned how well altering the coastal form on a large scale fits the philosophy of WWNP.

A much smaller scale approach has been trialled at Poole in Dorset where 30,000m³ of sand was placed in the nearshore area and monitored to ascertain if it would be transported to the beach. After 14 months, little onshore movement had occurred

(Poole Borough Council 2016). Monitoring results suggested that larger quantities of sand placed closer inshore might be more effective.

5.5.4 Effectiveness/performance

A common difficulty is obtaining recharge material that is acceptable from the viewpoint of the natural environment, amenity and aesthetics, and that can be expected to produce a long-lasting improvement in the beach (CIRIA 2010). On most beaches, sorting of sediment by waves and tides will have occurred over many years. As a result, the natural material size and grading at a particular site will provide a good guide to what material can be expected to remain on the beach, rather than being lost offshore.

The gradient of a stable beach is related to the size of particles found within it. In general, larger particle sizes will have a greater stable gradient (Table 5.8).

Sediment type	Median sediment	Mean beach gradient		
	size (D50) (mm)	From	То	
Sand	0.2	1:50	1:100	
	0.3	1:25	1:50	
	0.5	1:20	1:40	
Shingle	5.0	1:8	1:15	
	10.0	1:7	1:12	
	35.0	1:4	1:8	

 Table 5.8
 Typical beach slopes for various mean sediment sizes

Source: CIRIA (2010)

Optimising FCERM function is often achieved using relatively coarse grade beach material, which can achieve a steeper angle of repose. This can be at odds with the availability of material from dredging, which often has a wide grading curve with a high proportion of finer material. Use of as-dredged material can have a variety of adverse consequences to FCERM management, and the nature conservation value and amenity of the beach. Beach material containing a high fine content may be prone to erosion by cliffing rather than behaving dynamically under wave action. It may also promote the growth of non-characteristic 'weedy' vegetation. Possible solutions are to screen dredged material, or to find and use find alternative sources, although both are likely to significantly increase the expense of beach recharge.

Land-based sources (for example, quarries) enable particle size distribution to be controlled but material may be angular, which affects FCERM performance and the amenity value of the beach. The use of high quality quarried aggregates for beach recharge is likely to be very expensive and of doubtful sustainability.

In some cases, the lithology of recharge material may be a consideration, for example, within geological SSSIs and geological World Heritage Sites (of which the UK has 2). Conserving the integrity of these sites may require beach recharge to be undertaken using material of similar origin and lithology. An alternative, which has sometimes been proposed by Natural England where similar material cannot be obtained, is that material of very different lithology and appearance be used so that is can be recognised within the system subsequently and not confused with indigenous material.

5.5.5 Key case studies

Lincshore

One of the largest beach recharge programmes in the UK is between Mablethorpe and Skegness on the Lincolnshire coast. The frontage is ~24km long, mostly low-lying, with predominantly agricultural land use, Housing below the level of the high tide level is at risk of flooding, and is subject to a rising sea level and falling sediment supply. Much of the frontage from Mablethorpe to Skegness has been engineered and hard defence structures such as seawalls with promenades, revetments and rock amour are present (Environment Agency, undated c).

The coast in this area has been subject to long-term historic erosion and general retreat of the beaches. Where the beaches are narrow and backed by hard defences, this may compound coastal squeeze. This seems especially apparent at the heavily defended promenades such as Vickers Point and Ingoldmells Point. The whole Lincshore section is experiencing year-on-year erosion and, in the past, storm conditions have led to the exposure of the underlying clay platform.

Management activities are set out in the Saltfleet to Gibraltar Point Strategy. In 1994, a nourishment scheme (Lincshore) began to replenish lost sediment along the frontage and continues to do so to date. Renourishment occurs in early spring, along most of the frontage with a particular focus on recurring 'hot spots'. A pipeline is buried under the upper beach to assist this renourishment. The scheme maintains the standard of protection against a 1 in 200 chance (0.5%) of flooding in any one year by covering the clay with sand, reducing the risk of tidal flooding to more than 30,000 homes and businesses, 19,000 static caravans and 35,000ha of land that are at risk of flooding.

Poole Nearshore Replenishment Trial

The experiments by the Borough of Poole Council, with support from the Environment Agency and the Channel Coastal Observatory, were the first trial of the nearshore (subtidal) replenishment method in the UK when 30,000m³ of material dredged from Poole Harbour was deposited on the seabed ~350m offshore of Shore Road beach, Canford Cliffs, on 14 February 2015 (Poole Borough Council 2016). With this method, sediment is deposited typically on an offshore sand bar, and over time waves transport the sediment to the beach face. The technique is expected to have several advantages over conventional methods of beach renourishment used in the UK, including the potential of being both more economically and environmentally sustainable.

As a condition of the Marine Management Organisation (MMO) deposition licence, an extensive monitoring programme was required to assess the behaviour of the deposited material. An important secondary requirement was to determine the potential impacts of the nearshore deposition on sensitive/protected marine features, particularly given the proximity of the Poole Rocks Marine Conservation Zone.

From a scientific and engineering point of view, the primary purpose of the monitoring was to establish whether small volumes of material deposited in the nearshore region can effectively trickle charge the beach in sufficient quantities to replace the more traditional beach replenishment method.

To track the dispersal of the deposited material, the Poole Bay Nearshore Replenishment Trial monitoring programme involved:

- beach (topographic) surveys
- swath bathymetry (multibeam) surveys

- seabed tracer study
- waves, currents and turbidity measurements
- tidal monitoring
- silt monitoring (by the University of Southampton, commissioned separately by the Borough of Poole Council)

The main findings of this monitoring were as follows.

- Some 14 months after deposition, the mounds of sediment remained distinct features, approximately 2m high. The sediment had remained in situ, with only ~1,000m³ (~3%) net loss since deposition. Such small net volumes of sediment change, however, are difficult to identify even from high precision bathymetric and topographic surveys.
- Although little net volume change has occurred, the mounds have shown signs of shoreward translation by about 10m in the manner of an offshore bar.
- Under the right hydrodynamic conditions, sediment can be transported quickly from 5m Chart Datum (CD) to the beach, although the majority of sediment is transported alongshore. During storm conditions, longshore currents are generally directed towards the north-east.
- No detrimental effect on Poole Rocks Marine Conservation Zone was observed.

It was concluded that any future deposition should be placed at ~5m CD, rather than further offshore at ~8 m CD, where it would be more likely to be transported parallel to the coast than onshore. A larger volume of recharge would be more likely to lead to meaningful results.

North Norfolk sandscaping (proposed)

A feasibility study by Royal HaskoningDHV (2015a) presented an analysis of the potential for sandscaping on the north Norfolk coast between Mundesley and Cart Gap (Happisburgh). The report noted that the characteristic features of this coast such as nesses, spits and subtle changes in coastal orientation influence, regulate and supply sediment over larger areas in a natural way. The nearshore and offshore zones are characterised by shoals and sandbanks, which again influence wave and sediment transport processes. In many ways, these features demonstrate the basic sandscaping concept in introducing a sediment feature that is able to influence the way in which the coast functions in relation to the needs of coastal management. The feasibility study built on this concept by considered these natural features and processes, and looking at ways in which this naturally functioning system could be enhanced.

The objectives of the feasibility study for a sandscaping approach to the north Norfolk coast included examining the potential benefits, recognising the uncertainties, and outlining the costs involved so as to provide a more level playing field compared with more traditional approaches to defence and coastal management.

The sandscaping concept opens up different possibilities for delivering the intent of the SMP2 for the area, changing the timescales of policies and the attitude towards adaptation while still building on the same underlying understanding and principles for sustainable management.

The analysis by Royal HaskoningDHV assessed the quantity of sandscaping nourishment required to achieve FCERM outcomes at least equivalent to the current SMP policy for the area. The study also identified where such an approach to coastal management would enable additional benefits to be realised in terms of:

- enabling an alternative approach to coastal protection
- enhancing the natural environment
- providing socioeconomic benefits such as improved quality of life, additional amenity value and potential for inward investment in the area

The study defined the following objectives (Royal HaskoningDHV 2015a):

- Functional performance objectives:
 - To develop proposals for sandscaping based on a thorough interrogation of evidence and a strong understanding of the coastal processes so that the impacts on all constraints and the risk of unintended consequences are minimised
 - To provide coast protection to the Bacton Gas Terminal, a nationally important critical infrastructure with a direct asset value of over £200 million
 - To reduce the risk of erosion and defence failure to the Bacton, Walcott and Ostend frontage in the short term (0–20 years) as a minimum, in accordance with SMP policy
 - To ensure sandscaping nourishment is of sufficient size to influence the coastal processes so that beach levels are maintained at key locations, while also maintaining and potentially enhancing sediment drift along the coast
 - To optimise the effective life of any nourishment programme, considering the associated timescales for downdrift impacts (for example, when will upstream nourishment reach Cart Gap)
 - To optimise the balance between the presence of a beach that provides protection against erosion and the reduction of sediment supply from the cliffs
 - To minimise impacts on, and where possible enhance, the natural environment including foreshore and nearshore habitats, geological features and fisheries
- Opportunity benefits:
 - To deliver risk management efficiencies by offsetting expenditure incurred or expected elsewhere, for example, Cart Gap (nourishment programme) and Mundesley (proposed Hold the Line scheme)
 - To enable an alternative approach to adaptation to coastal change that manages and balances the conflicting issues for coastal management on this part of the north Norfolk coast
 - To maximise the positive natural environment benefits of the scheme, for example, creation of intertidal habitat, reduced habitat disturbance through reduced frequency of nourishment at Sea Palling
 - To maximise the potential benefits to local businesses and for inward investment for economic growth of the area

- To provide positive benefits to local communities through improved erosion risk management (with associated safety and welfare impacts), and creating the potential for improving quality of life in other ways

The study proposed the following volumes of recharge:

- 5,156,250m³ Phase 1 nourishment
- 4,125,000m³ Phase 2 nourishment, after 25 years

Economic appraisal showed that the benefit–cost of sandscaping is in the range 2.28 to 3.20 for the Bacton Gas Terminal and in the range 2.02 to 2.72 for Bacton to Walcott, compared with 3.03 for the current SMP management policy. (The actual amounts of benefits and costs are not clear as the units are not stated in Royal HaskoningDHV 2015a). The benefit–cost ratios for sandscaping are sensitive to the costs of the recharge and to inclusion of the benefits associated with habitat creation and improved amenity value (increased visitor numbers). The feasibility study suggested that, if the economic value of these is included, sandscaping becomes the most economically attractive option (Royal HaskoningDHV 2015a).

5.5.6 Funding

As a FCERM measure, beach recharge is usually funded by Defra grant-in-aid. Because beach recharge frequently has amenity benefits, other supplementary sources of funding may also be available through partnership funding. For example, a scheme at Weston-super-Mare was co-funded by North Somerset District Council and Defra, while seafront enhancements were funded by the South West of England Regional Development Agency (CIRI, 2010).

Flood defence grant-in-aid is allocated based on demonstrating a scheme is cost beneficial and reduces risk to people and property. Sandscaping is so expensive that initial assessment by the Environment Agency indicates it would be very hard to justify this in the UK.

5.5.7 Design, management and maintenance

Beach recharge usually has to be repeated at frequent intervals if the standard of coastal and flood risk management is to be maintained. Natural processes generally transport recharge material both longshore and cross-shore, so it becomes redistributed and lost to the part of the system where FCERM is needed. Frequently, structures such as groynes or breakwaters are constructed as well as recharge to contribute to retaining material where it is needed. These work essentially by interfering with natural processes and hence are not included in this research. Another way to minimise maintenance and management requirements is using the sandscaping approach discussed above.

5.5.8 Other requirements

Beach usually recharge involves the deposition of material above low water mark and below high water mark. As such, it requires planning consent from the local council and a marine license from the MMO. In general, applications for these consents will need to be accompanied by an Environment Statement under the Town and Country Planning (Environmental Impact Assessment) Regulations and the Marine Works (Environmental Impact Assessment) Regulations. If recharge is undertaken below the low water mark (as in the Dutch sand engine example), then it falls under the jurisdiction of the MMO only.

5.5.9 Research gaps

Beach recharge is a well-established technique, for which FCERM benefits frequently provide robust economic justification. When appropriately managed using suitable source material, it can deliver multiple benefits such as to recreational amenity, landscape and nature conservation. However, it also has the potential to disrupt habitats at the material source and receptor locations. The sustainability of this approach depends on the continuing availability of beach material.

Sandscaping is a large-scale version of beach recharge, which has yet to be undertaken in the UK. Feasibility studies to date indicate that it could have additional, albeit diffuse, benefits to the environment. Any economic justification for sandscaping, rather than traditional beach recharge techniques are likely to depend on assigning an economic value to these multiple benefits. The cost-effectiveness of this approach in the UK, where FCERM funding differs from the Netherlands, has not been demonstrated. The intention of sandscaping is to modify coastal processes and so it is debatable to what extent it represents WWNP. More objective analysis is needed for further consideration to be given to adopting sandscaping.

Important research gaps include the following.

- There is uncertainty over how recharge material of different grading to indigenous material will behave on beach as it becomes sorted by natural processes.
- The many uncertainties relating to sandscaping include the impact on coastal processes and economic viability. Techniques and data availability for ecosystem services valuation need to be improved if sandscaping is to be seen as a viable approach.

5.6 Sediment bypassing

5.6.1 Understanding the science

Sediment bypassing is the artificial movement of beach material around a harbour mouth, groyne or other obstruction in the same direction as longshore drift. It is effectively mitigation for the interruption of longshore drift caused by an artificial obstruction. Sediment bypassing involves moving material from an area of accumulation to an area of erosion. As such, sediment bypassing (unlike sediment recycling, which moves material opposite to the direction of longshore drift) constitutes WWNP.

Essentially, sediment bypassing is a way of supplying material to a beach where there is a deficit so as to sustain the beach to provide the required FCERM standard. It is one potential source of material; other sources include recycling and recharge.

The concept of sediment bypassing was originally developed in response to the problems associated with tidal inlets (CIRIA 2010). These include excessive accretion updrift of an inlet jetty and unwanted sedimentation seawards of the inlet, and often acute erosion downdrift as a result of sediment being trapped updrift.

There are a number of techniques for sediment bypassing:

- **Mechanical bypassing** physical trucking of material from one side of the obstruction to the other. In built-up areas, this may have significant effects in terms of traffic, cost, public amenity and environmental considerations. In theory, marine plant (barges) could be used to avoid the road traffic impact. Another alternative could be fixed plant such as a conveyer belt to move beach material over the obstruction.
- **Hydraulic bypassing** using fixed, mobile or semi-mobile bypassing plant to transport sand as a slurry via suction pipes, or using a centrifugal pump to a discharge pipeline.
- Seabed fluidisation use of jet pumps or other means of bed fluidisation by forcing water through a nozzle under pressure to create a sand–water slurry that can readily be pumped.

Other bypassing methods such as sediment traps and weir jetties have not been tried in the UK.

Few areas in the UK have sufficiently persistent and sustained rapid shoreline advance and accompanying downdrift erosion to make bypassing with fixed plant an economic proposition. However, sediment bypassing with mechanical plant (such as excavators and trucks) may be economically effective in a wider range of locations, using similar techniques to those of sediment recycling. This approach is flexible and allows for changes in quantity or even drift direction. Areas that may be suitable for this type of management include approaches to ports, navigation channels and areas of accumulation in harbour shoals. Bypassing operations can be included as part of a capital or maintenance dredging programme to transfer material from an area of siltation in a harbour approach channel to downdrift areas at relatively low cost (CIRIA 2010).

5.6.2 Confidence in the science

Sediment bypassing operations have many potential advantages in reducing the longshore environmental effects of beach control structures. By maintaining the supply of material along the shore, sediment bypassing ensures that downdrift erosion is reduced. Bypassing can also help to avoid the build-up of material to the updrift side of a structure, which can be detrimental to navigation or other interests. Sediment added to a cell can be placed where the greatest shortage exists, or alternatively added as far updrift as possible so longshore drift then ensures it benefits the whole cell (Dornbusch, 2016, personal communication).

Most examples of long-term beach bypassing operations in the UK are on the Sussex coast, where the longshore drift direction from west to east is interrupted by a number of natural and man-made harbours and associated anthropogenic structures.

The science behind beach bypassing is well-understood. Where bypassing is round an artificial harbour or groyne, it is effectively mitigation for human interference with longshore drift. Where bypassing is around a natural harbour or obstruction, it overcomes a barrier to natural transport and is still classed as WWNP as material is moved in the same direction as longshore drift.

In the examples discussed below from the Sussex coast, bypassing beach material does not provide sufficient material to fulfil the whole FCERM requirement immediately downdrift. In the case of Pevensey Bay and Pagham Harbour, it is understood that about one-third of the downdrift requirement is provided by bypassing. Other sources of material are therefore needed as well. Given the inherent variability in the volume of material moved by longshore drift and the fact that material is also moved cross-shore

(and therefore lost to the nearshore cell), it is likely to be inevitable that any bypassing scheme will only form part of an FCERM solution to the downdrift frontage.

5.6.3 Metrics of cost–benefit and multiple benefits

Beach bypassing is an established technique in beach management and is usually justified on the basis of traditional economic appraisal. It may be primarily driven by FCERM considerations or by maintaining navigation (where in the absence of bypassing material would otherwise block a harbour channel), or a combination of the two.

Multiple benefits from recycling at Pagham Harbour (see case study 63) include:

- maintaining the standard of defence (1 in 200 years) to built properties on the frontage
- obtaining one-third of the required material through bypassing (hence WWNP)
- achieving the aims of the approved strategy for the frontage
- avoiding significant impact on environmental designations in the area
- · lowest carbon usage score of the options
- support from all statutory consultees, other stakeholders and the community

Pevensey Bay frontage in East Sussex has been carried out since 2007 by dredging material previously deposited offshore following maintenance dredging of Sovereign Harbour (Pevensey Coastal Defence 2015).

An economic appraisal is not available for the Pevensey Bay scheme in East Sussex where mechanical bypassing of material dredged from Sovereign Harbour occurs. The scheme is managed under a public–private partnership (PPP) by Pevensey Coastal Defence Limited. Multiple benefits from this scheme include the following.

- Sovereign Harbour development was consented partly on the understanding that sediment which was no longer able to drift freely into Pevensey Bay would be mechanically bypassed round the harbour by road.
- WWNP favours bypassing into Pevensey Bay.
- It is estimated that bypassing to Pevensey is ~£2 per m³ cheaper than recycling to Eastbourne Pier and Bandstand (shorter recycles would be more cost-effective).
- The scheme has reduced social impacts compared with recycling. There is a much higher density of population, tourism and businesses along Eastbourne's Royal Parade and Grand Parade than along the A259 to Sovereign Harbour North and therefore the impact of traffic to Eastbourne be greater than trucking the material to Pevensey.
- Responsiveness is improved. The South West Beach at Sovereign Harbour is a highly volatile section of coast, and sediment can build up to a point where it should be removed in days. Pevensey Coastal Defence is best placed to initiate bypassing works at short notice.

In general, beach bypassing may be expected to have a negative impact on nature conservation interests where the material is removed from (reduction in habitat) and a

positive impact (similar to those of beach recharge) at the location where it is deposited (gain in habitat). Impacts on the donor site may be negligible if material has accreted over a short time and has no associated nature conservation interests. There are also potential negative implications from habitat disturbance involved in moving and placing the material.

5.6.4 Effectiveness/performance

Beach bypassing is flexible and can provide a high standard of FCERM provided sufficient material is available. In practice, bypassed material is often supplemented by recharge material from other sources such as dredging.

At Pagham Harbour, bypassing provides about one-third of the beach material needed to achieve a 1 in 200 year defence standard against breach.

At Pevensey Bay, bypassing provides about one-third of the beach material needed to achieve a 1 in 400 year defence standard against breach.

5.6.5 Key case studies

Most examples of long-term beach bypassing operations in the UK are on the Sussex coast, where the longshore drift direction from west to east is interrupted by a number of natural and man-made harbours and associated anthropogenic structures. The 3 examples described below are from Pevensey Bay, Shoreham Harbour and Pagham Harbour.

In all these cases, there is a need for a sediment feed downdrift of the interruption caused by the harbour entrances and so bypassing fulfils a FCERM function as well as a navigation purpose. No economic assessments on the relative benefits of each purpose are available.

Alternatively, the need for sediment for FCERM could be satisfied by beach recycling or recharge. In the case of Shoreham, recycling beach material back from Brighton Marina is actually slightly cheaper because the distance of transport is shorter. In the case of Pevensey, bypassing is the cheapest of the 3 options. However, recycling has the benefit of not losing the material from the system. For example at Sovereign Harbour, the dredge spoil is deposited kilometres offshore, and if material accumulating west of the harbour arm were deposited there, it would mean a loss of shingle from the system, which has negative FCERM implications.

Pevensey Bay

The Pevensey Bay scheme³⁵ is managed by Pevensey Coastal Defence Limited under a 25 year (2000 to 2025) PPP contract with the Environment Agency based on fixed annual payments. However, the contract is flexible enough to accommodate change.

Eastbourne and Pevensey Bay are at the updrift end of sediment cell 4b. The general longshore transport direction moves beach material along the Eastbourne frontage towards Sovereign Harbour at Pevensey. Historically this material would have found its way into Pevensey Bay, but now accumulates in the area between Langney outfall and the southern harbour arm. As the process of longshore transport continues, the beach against the harbour arm would keep growing and eventually spill round the harbour arm and into the entrance channel. To avoid this and to aid the beach material on its way eastwards, surplus shingle (5,000–15,000m³ per year) is taken from the beach and

³⁵ <u>http://www.pevensey-bay.co.uk/</u>

transferred – or bypassed – round the harbour by lorry. It is placed either in a stockpile behind the north harbour's rock revetment or added directly to the scour hole at the eastern end of the revetment. The stockpile acts as an emergency source of material for the winter months, when it is often too rough for dredgers to operate for extended periods. Because of constraints imposed by Eastbourne Borough Council, the transfer operation only takes place between 1 October and 31 March each winter. This material forms part of the FCERM programme for Pevensey Bay, which is provided by Pevensey Coastal Defence under the 25-year contract to provide a 1 in 400 year standard of defence against breach.

The sediment deficit of 30,000m³ on the Pevensey frontage is replenished in 2 ways:

- Recharge from offshore dredged material from Owers Bank (near Littlehampton) is rainbowed onto the beach at the western end of the frontage; typically around 21,000m³ are recharged annually
- Bypassing of material accumulated against the southern harbour arm around Sovereign Harbour by road lorries between 4,000m³ and 25,000m³ are moved each year with an annual average in recent years of 9,000m³.

Contractually, Pevensey Coastal Defence expects to receive on average 5,000m³ of shingle annually, bypassed by road round Sovereign Harbour and added to the beach on the north side of the harbour. The contract allows Pevensey Coastal Defence to take 5,000m³ of material for bypassing at no charge – though it pays transport costs. For material over this amount, Pevensey Coastal Defence pays the difference between the cost of importing material from offshore and the cost of bypassing Sovereign Harbour. This value has been index linked since 2003 (RPIX³⁶) and was £16.05 per m³ until June 2017. While there is only a marginal financial incentive for Pevensey Coastal Defence to use bypassed material, it is of higher quality (lower fine content) and is available at short notice when dredgers are not available or able to operate. Pevensey Coastal Defence is contractually required to use bypassed material when instructed to do so.

Pevensey Coastal Defence (Thomas, 2016, personal communication) has stated that ideally the sub-cell should be managed by feeding the updrift end and managing the sediment as it drifts east by means of recycling, reprofiling and bypassing. This inevitably leads to a build-up over time at the downdrift end. It has not yet been established whether this can be used as source material for recycling by sea or in some other way. There is a net loss of sediment above 0m Ordnance Datum (OD) from Eastbourne and Pevensey Bay estimated at 40,000m³ annually. Pevensey Coastal Defence considers that additional recharge to offset this loss should then be added downdrift of Sovereign Harbour, enabling the remainder of the sub-cell to be generally managed without further shingle input.

Shoreham Harbour

The natural process of shingle movement along the Sussex coast results in the buildup of ~15,000m³ of shingle annually at the east end of Shoreham Beach, where its progress is blocked by the west breakwater. This blocking of further eastwards shingle movement by the harbour entrance breakwaters causes erosion problems on Southwick Beach. Shoreham Port has powers under the Harbour Act to undertake a programme of shingle bypassing, whereby shingle is excavated from the west side of the entrance and transported and placed on the east side. This process is split evenly

³⁶ Retail Price Index all items excluding mortgage interest

between spring and autumn campaigns, and care is taken to minimise the impact of the operation on local residents.

Studies have shown that, as a result of shingle bypassing, beach volumes in the area have improved and stabilised. The shingle bypassing operation is supported by the South Downs SMP and by the 2 coastal defence strategies that define the need for sea defence improvements between the River Arun and Brighton Marina.³⁷ Shingle bypassing has been undertaken since 1993, when 10,000 tonnes were moved. Between 2007 and 2011, volumes ranged from 22,000 tonnes to 30,000 tonnes (Halcrow 2012).

Pagham Harbour

An increase in the rate of shingle accumulation at Church Norton Spit is thought to reduce shingle feed onto the Pagham Beach frontage and might have been responsible for historical erosion along parts of this frontage. As a result, 76 residential and commercial properties are at a direct increased risk of erosion. In 2009, best available estimates predicted continued evolution over the medium and longer term, placing over 300 residential and commercial properties located within the Pagham conurbation at increased flood and coastal erosion risk. It was also predicted that in the following 5 years, there would be an approximate net loss of 15,000m³ of shingle increasing to an estimated deficit of 30,000m³ in year 10 and 150,000m³ by year 50 from the Pagham Beach frontage. These figures did not include any loss from storm events that may occur within the same period, nor the additional risk and vulnerability to storm events for a depleted frontage.

The proposed solution (Arun District Council 2009) involved a combination of bypassing 10,000m³ of shingle round Pagham Harbour entrance and recharge of a further 20,000m³ from offshore. A combination of bypassing and recharge was decided on because Natural England objected to bypassing of more than 10,000m³ and because the combination option gives the lowest cost and best benefit–cost ratio.

5.6.6 Funding

Beach bypassing with the objective of maintaining FCERM downdrift will usually be eligible for funding by Defra grant-in-aid.

Where bypassing also has a navigation benefit (for example, maintaining a dredged channel to a port or harbour), financial contributions may be made by the port or harbour authority. In many cases, the port authority may be best placed to undertake the bypassing operation, as at Shoreham. A contribution may then be payable from the flood defence authority to the port authority to reflect the FCERM value of bypassed material.

5.6.7 Design, management and maintenance

Beach bypassing usually needs to be carried out on an ongoing basis, for example, annually. The quantities to be bypassed may either be set in advance or derived from monitoring of the volume of material available or required. Where the requirement is to maintain a given standard of defence at the downdrift (receiving) location, provision may have to be made for alternative sources of material to be used if there is insufficient at the updrift (donor) location.

³⁷ <u>http://www.shoreham-port.co.uk/Shingle-Bypassing</u>

At Pevensey, Pevensey Coastal Defence performs 2 other beach management activities in addition to bypassing and recharge.

- Dredged material from the navigation channel adjacent to the harbour entrance is deposited below low water springs (this material is finer than material taken from the beach). This dredging is done by harbour owners Premier Marinas and not by Pevensey Coastal Defence. Pevensey Coastal Defence works with Premier Marinas to try and ensure suitable sediments are beneficially reused. About 40,000m³ have been reused in the last 2 years rather than being dumped at sea.
- Recycling occurs from the eastern end of the frontage westwards. This is done piecemeal, moving material various distances but rarely the full length of the frontage. Recycling volumes average about 100,000m³ annually.

Other flood risk management measures carried out by Pevensey Coastal Defence on the frontage are beach profiling, constructing/maintaining retaining walls to hold beach material in place, and maintaining about 7 groynes. The remaining 150 groynes on the frontage are not being maintained and had already deteriorated to a poor standard before the project started.

Pevensey Coastal Defence has also adapted management of the Pevensey frontage in the light of experience over the 16.5 years the contract has been running.

- Beach recharge material was originally pumped on to the beach and then distributed by bulldozers. It is now pumped to the western end of the frontage and natural processes are allowed to distribute it along the frontage.
- A convoy system has been introduced for lorries bypassing beach material to reduce the duration of impact on local residents.
- Sand and silt accumulated in Sovereign Harbour has been used on the lower shore of the Pevensey frontage (below 0m OD) to increase the stability of the beach; 17,000m³ were used in 2014 and 32,000m³ in 2015. This is not part of the contract (which only counts beach material above 0m OD) but it offsets losses that are occurring and increases the stability of the frontage at minimal cost to Pevensey Coastal Defence.
- Sea level rise since 2000 has been at only half the rate predicted in the contract. However, storm events have been more severe (especially in 2013 to 2014), leading to loss of shingle and the need for additional replenishment.

Beach profiles are monitored by Pevensey Coastal Defence monthly and by Channel Coastal Observatory for the Environment Agency 2–3 times per year.

5.6.8 Other requirements

As beach bypassing involves excavation and placement of material above low water mark and below high water mark, planning consent and a marine licence are usually needed. As shingle habitats and movement of shingle by road are environmentally sensitive, an Environmental Statement is likely to be required to accompany consent applications. Good liaison with stakeholders and the local community are an important element of both the Pevensey Bay and Pagham Harbour bypassing schemes, with use made of websites and public meetings to explain the projects.

5.6.9 Research gaps

Beach bypassing is a well-established technique, providing FCERM and potentially navigation benefits. These dual benefits can provide increased economic justification for bypassing rather than other forms of beach replenishment such as recharge from offshore or recycling. However, there may be negative environmental implications to beach habitats at the source location and to residential amenity where material is moved by lorry. In practice, bypassing is only applicable to a relatively small number of situations. Furthermore, it is likely to represent only part of the solution to replenishing a deficit of material downdrift. Other additional sources of material such as dredged recharge are usually needed.

References

ABBE, T.B. AND MONTGOMERY, D.R., 1996. Large woody debris jams, channel hydraulics and habitat formation in large rivers. *Regulated Rivers: Research and Management*, 12 (2-3), 201-221.

ABBE, T.B. AND MONTGOMERY, D.R., 2003. Patterns and processes of wood debris accumulation in the Queets river basin, Washington. *Geomorphology*, 51 (1-3), 81-107.

ABEL, S., HOPKINSON, L. and HESSION, W., 2016. Hydraulic and Physical Structure of Runs and Glides Following Stream Restoration. *River Research and Applications*.

ÅBERG, E.U. AND TAPSELL, S., 2013. Revisiting the River Skerne: the long-term social benefits of river rehabilitation. *Landscape and Urban Planning*, 113, 94-103.

ABPMER, 2014. Online Marine Registry (OMReg). Database of international shoreline adaptation projects (latest update 30 July 2014) [online]. Southampton: ABPmer. Available from: <u>http://www.omreg.net</u> [Accessed June 2016].

ABPMER, 2015. *The cost of managed realignment* [online]. Southampton: ABPmer. Available from: www.abpmer.co.uk/buzz/the-cost-of-managed-realignment/ [Accessed 4 July 2017].

ABRISQUETA, J.M., PLANA, V., MOUNZER, O.H., MENDEZ, J. AND RUIZ-SANCHEZ, M.C., 2007. Effects of soil tillage on runoff generation in a Mediterranean apricot orchard. *Agricultural Water Management*, 93 (1), 11-18.

ACREMAN, M. AND HOLDEN, J., 2013. How wetlands affect floods. *Wetlands*, 33 (5), 773-786.

ACREMAN, M.C., RIDDINGTON, R. AND BOOKER, D.J., 2003. Hydrological impacts of floodplain restoration: a case study of the River Cherwell, UK. *Hydrology and Earth System Sciences*, 7, 75-85.

ACREMAN, M.C., FISHER, J., STRATFORD, C.J., MOULD, D.J. AND MOUNTFORD, J.O., 2007. Hydrological science and wetland restoration: some case studies from Europe. *Hydrology and Earth System Sciences*, 11 (1), 158-169.

ACREMAN, M.C., HARDING, R.J., LLOYD, C., MCNAMARA, N.P., MOUNTFORD, J.O., MOULD, D.J., PURSE, B.V., HEARD, M.S., STRATFORD, C.J. AND DURY, S.J., 2011. Trade-off in ecosystem services of the Somerset Levels and Moors wetlands. *Hydrological Sciences Journal-Journal Des Sciences Hydrologiques*, 56 (8), 1543-1565.

ADA, 2015. *The value of water level management: economy*. Kenilworth, Warwickshire. Association of Drainage Authorities. Available from: <u>http://www.ada.org.uk/key-topics/value-of-water-level-management/</u> [Accessed 4 July 2017].

ADAPTATION SUB-COMMITTEE, 2013. *Managing the land in a changing climate. Adaptation Sub-Committee progress report 2013.* London: Committee on Climate Change.

ADAMS, W.M. AND PERROW, M.R., 1999. Scientific and institutional constraints on the restoration of European floodplains. In *Floodplains: Interdisciplinary Approaches* (ed. S.B. Marriott and J. Alexander), Special Publications163, pp. 89-97. London: Geological Society.

ADAS, 2002. *Development of a database of agricultural drainage*. Defra ES0111. London: Department for Environment, Food and Rural Affairs.

ADDY, S. AND WILKINSON, M.E., 2016. An assessment of engineered log jam structures in response to a flood event in an upland gravel-bed river. *Earth Surface Processes and Landforms*, 41 (12), 1658-1670.

ADDY, S., WILKINSON, M.E. AND COOKSLEY, S., 2016. Geomorphic changes and hydrological responses to the 2015 'Storm Frank' flood event at a river restoration site on the upper River Dee. Presentation to 4th British Hydrological Society International Conference (Cranfield, September 2016).

AHDB, 2015. *Field drainage guide: principles, installation and maintenance.* Stoneleigh Park, Warwickshire: Agricultural & Horticultural Development Board.

ALILA, Y., KURAŚ, P.K., SCHNORBUS, M. AND HUDSON, R., 2009. Forests and floods: a new paradigm sheds light on age-old controversies. *Water Resources Research*, 45, article W08416.

ALVARENGA, C.A., MELLO, C.R., MELLO, J.M. AND VIOLA, M.R., 2011. Spatial continuity of the saturated hydraulic conductivity in soils of the Alto Rio Grande basin, MG. *Revista Brasileira de Ciência do Solo*, 35 (5), 1745-1757.

ANDERSON, B.G., RUTHERFURD, I.D. AND WESTERN, A.W., 2006. An analysis of the influence of riparian vegetation on the propagation of flood waves. *Environmental Modelling and Software*, 21 (9), 1290-1296.

ANDERSON, H.W., 1969. Snowpack management. In *Snow*, proceedings of Oregon State University Water Resource Institute Seminar WR 011.69, pp. 27-40. Corvallis, OR: Oregon State University

ANDERSON, H.W., 1970. Storage and delivery of rainfall and snowmelt as related to forest environment. In *Proceedings of Third Microclimate Symposium* (Seebe, Alberta, Canada, September 1969), edited by J.M. Powell and C.F. Nolasco, pp. 51-67. Calgary, Alberta: Canadian Forestry Service, Forest Research Laboratory.

ANDERSON, H.W., HOOVER, M.D., REINHART, K.G., 1976. Forests and water: *Effects of forest management on floods, sedimentation, and water supply.* General Technical Report PSW-18/1976. Berkeley, California: Forest Service, US Department of Agriculture.

ANDERSON, N.H., SEDELL, J.R., ROBERTS, L.M. and TRISKA, F.J., 1978. Role of aquatic invertebrates in processing of wood debris in coniferous forest streams. *American Midland Naturalist*, 100, 64-82.

ARCHER, D., 2003. Scale effects on the hydrological impact of upland afforestation and drainage using indices of flow variability: the River Irthing, England. *Hydrology and Earth System Sciences Discussions*, 7 (3), 325-338.

ARCHER, D. AND NEWSON, M., 2002. The use of indices of flow variability in assessing the hydrological and instream habitat impacts of upland afforestation and drainage. *Journal of Hydrology*, 268 (1-4), 244-258.

ARCHER, N.A.L., BONELL, M., COLES, N., MACDONALD, A.M., AUTON, C.A. AND STEVENSON, R., 2013. Soil characteristics and landcover relationships on soil hydraulic conductivity at a hillslope scale: a view towards local flood management. *Journal of Hydrology*, 497, 208-222.

ARMBRUSTER, M., SEEGERT, J. AND FEGER, K.H., 2004. Effects of changes in tree species composition on water flow dynamics. Model applications and their limitations. *Plant and Soil*, 264 (1-2), 13-24.

ARMSTRONG, A.C. AND HARRIS, G.L., 1996. Movement of water and solutes from agricultural land: the effects of artificial drainage. In: *Advances in Hillslope Processes* (ed. M.G. Anderson and S.M. Brooks), Volume 1, pp. 187-211. Chichester: Wiley.

ARNOLD, J.G., ALLEN, P.M. AND BERNHARDT, G., 1993. A comprehensive surfacegroundwater flow model. *Journal of Hydrology*, 142 (1-4), 47-69.

ARORA, K., MICKELSON, S. K., HELMERS, M. J. AND BAKER, J. L., 2010. Review of pesticide retention processes occurring in buffer strips receiving agricultural runoff. *Journal of the American Water Resources Association*, 46(3), 618-647.

ARUN DISTRICT COUNCIL, 2009. *Pagham Spit removal works. Project Appraisal Report.* Littlehampton, Sussex: Arun District Council.

AYRES, A., GERDES, H., GOELLER, B., LAGO, M., CATALINAS, M., GARCI CANTON, A., BROUWER, R., SHEREMET, O., VERMAAT, J., ANGELOPOULOS, N. & COWX, I., 2014. Inventory of river restoration measures: effects, costs and benefits. REFORM.

BALL, T., 2008. Management approaches to floodplain restoration and stakeholder engagement in the UK: a survey. *Ecohydrology and Hydrobiology*, 8 (2-4), 273-280.

BALLARD, C., 2011. *The role of physics-based models for simulating flood flow responses to rural land management scenarios*. Dept. Civil and Environmental Engineering, Imperial College London. PhD, 322 pp.

BALLARD, C.E., MCINTYRE, N. AND WHEATER, H.S., 2012. Effects of peatland drainage management on peak flows. *Hydrology and Earth System Sciences*, 16, 2299-2310.

BAPTIST, M.J., PENNING, W.E., DUEL, H., SMITS, A.J.M., GEERLING, G.W., VAN DER LEE, G.E.M. and VAN ALPHEN, J.S.L., 2004. Assessment of the effects of cyclic floodplain rejuvenation on flood levels and biodiversity along the Rhine river. *River Research and Applications*, 20 (3), 285-297.

BARBER, N., 2013. Sediment, nutrient and runoff management and mitigation in rural catchments. PhD thesis, Newcastle University.

BARBER, N.J. and QUINN, P.F., 2012. Mitigating diffuse water pollution from agriculture using soft-engineered runoff attenuation features. *Area*, 44 (4), 454-462.

BARBER, N., REANEY, S.M., BARKER, P.A., BENSKIN, C., BURKE, S., CLEASBY, W., HAYGARTH, P., JONCZYK, J., OWEN, G., SNELL, M.A., SURRIDGE, B. AND QUINN, P.F., 2016. *The Treatment Train approach to reducing non-point source pollution from agriculture*. Poster presented at the American Geophysical Union Fall General Assembly, abstract #H13D-1396.

BARR, C.J. AND GILLESPIE, M.K., 2000. Estimating hedgerow length and pattern characteristics in Great Britain using Countryside Survey data. *Journal of Environmental Management*, 60 (1), 23-32.

BARR, C.J., GILLESPIE, M.K., HOWARD, D.C., FULLER, R.M., HILL, R.A., GERARD, F.F., 2000. *Patterns of biodiversity in the landscape. ECOFACT Module 7a.* London: Department of the Environment, Transport and the Regions.

BATHURST, J.C., BIRKINSHAW, S.J., CISNEROS, F., FALLAS, J., IROUMÉ, A., ITURRASPE, R., GAVIÑO NOVILLO, M., URCIUOLO, A., ALVARADOB, A., COELLO, C., HUBERD, A., MIRANDA, M., RAMÍREZ M. AND SARANDÓN, R., 2011. Forest impact on floods due to extreme rainfall and snowmelt in four Latin American environments 2: Model analysis. *Journal of Hydrology*, 400 (3), 292-304. BAUDRY, J., BUNCE, R.G.H. AND BUREL, F. 2000. Hedgerows: an international perspective on their origin, function and management. *Journal of Environmental Management*, 60 (1), 7-22.

BEAUMONT, N.J., JONES, L., GARBUTT, A., HANSOM, J.D. AND TOBERMAN, M., 2013. The value of carbon sequestration and storage in coastal habitats. *Estuarine, Coastal and Shelf Science*, 137, 32-40

BEAVER SALMONID WORKING GROUP, 2015. *Final report of the Beaver Salmonid Working Group*. Inverness: Scottish Natural Heritage.

BECK, H.E., BRUIJNZEEL, L.A., VAN DIJK, A.I.J.M., MCVICAR, T.R., SCATENA, F.N. AND SCHELLEKENS, J., 2013. The impact of forest regeneration on streamflow in 12 mesoscale humid tropical catchments. *Hydrology and Earth System Sciences*, 17 (7), 2613-2635.

BEECHIE, T.J., SEAR, D.A., OLDEN, J.D., PESS, G.R., BUFFINGTON, J.M., MOIR, H., RONI, P. and POLLOCK, M.M., 2010. Process-based principles for restoring river ecosystems. *BioScience*, 60 (3), 209-222.

BEHARRY-BORG, N., SMART, J.C., TERMANSEN, M. AND HUBACEK, K., 2013. Evaluating farmers' likely participation in a payment programme for water quality protection in the UK uplands. *Regional Environmental Change*, 13 (3), 633-647.

BENNETT, S.J., WU, W., ALONSO, C.V. and WANG, S.S., 2008. Modeling fluvial response to in-stream woody vegetation: implications for stream corridor restoration. *Earth Surface Processes and Landforms*, 33 (6), 890-909.

BENITO, E., SANTIAGO, J.L., DE BLAS, E. AND VARELA M.E., 2003. Deforestation of water-repellent soils in Galicia (NW Spain) effects on surface runoff and erosion under simulated rainfall. *Earth Surface Processes and Landforms*, 28 (2), 145-155.

BENTON, T.G., VICKERY, J.A. AND WILSON, J.D., 2003. Farmland biodiversity: is habitat heterogeneity the key? *Trends in Ecology and Evolution*, 18 (4), 182-188.

BERTOLDI, W., GURNELL, A.M. and WELBER, M., 2013. Wood recruitment and retention: the fate of eroded trees on a braided river explored using a combination of field and remotely-sensed data sources. *Geomorphology*, 180, 146-155.

BESCANSA, P., IMAZ, M.J., VIRTO, I., ENRIQUE, A. AND HOOGMOED, W.B., 2006. Soil water retention as affected by tillage and residue management in semiarid Spain. *Soil and Tillage Research*, 87 (1), 19-27.

BESCHTA, R.L., PYLES, M.R., SKAUGSET, A.E. AND SURFLEET, C.G., 2000. Peakflow responses to forest practices in the western cascades of Oregon, USA. *Journal of Hydrology*, 233 (1), 102-120.

BEST, A., ZHANG, L., MCMAHON, T., WESTERN, A., VERTESSY, R., 2003. A critical review of paired catchment studies with reference to seasonal flows and climatic variability. CSIRO Land and Water Technical Report 25/03. Canberra, Australia: Murray-Darling Basin Commission.

BETSON, R.P., 1964. What is watershed runoff? *Journal of Geophysical Research*, 69 (8), 1541-1552.

BEVEN, K. J., 2010. Preferential flows and travel time distributions: defining adequate hypothesis tests for hydrological process models. *Hydrological Processes*, 24(12), 1537-1547.

BEVEN, K., AND GERMANN, P., 1982. Macropores and water flow in soils. *Water resources research*, 18(5), 1311-1325.

BEVEN, K.J., HENDERSEN, D.E. AND, REEVES, A.D., 1993. Dispersion parameters for undisturbed partially saturated soil. *Journal of Hydrology*, 143 (1-2), 19-43.

BEVEN, K., ROMANOWICZ, R., YOUNG, P., HOLMAN, I., POSTHUMUS, H., MORRIS, J., ROSE S., O'CONNELL, P.E. AND EWEN, J., 2008. *FD2120: Analysis of historical data sets to look for impacts of land use and management change on flood generation*. London: Department for Environment, Food and Rural Affairs.

BIGGAR, J.W. AND NEILSEN, D.R., 1976. Spatial variability of the leaching characteristics of a field soil. *Water Resources Research*, 12 (1), 78-84.

BIGGS, J., STOATE, C., WILLIAMS P., BROWN, C., CASEY, A., DAVIES, S., GRIJALVO DIEGO, I., HAWCZAK, A., KIZUKA, T., MCGOFF, E. AND SZCZUR, J., 2014. *Water Friendly Farming. Results and practical implications of the first 3 years of the programme.* Oxford: Freshwater Habitats Trust; Fordingbridge: Game & Wildlife Conservation Trust.

BIGGS, J., STOATE, C., WILLIAMS P., BROWN, C., CASEY, A., DAVIES, S., GRIJALVO DIEGO, I., HAWCZAK, A., KIZUKA, T., MCGOFF, E., SZCZUR, J. AND VILLAMIZAR VELEZ, M., 2016. *Water Friendly Farming. Autumn 2016 update.* Oxford: Freshwater Habitats Trust; Leicester: game & Wildlife Conservation Trust.

BILBY, R.E., 1984. Removal of woody debris may affect stream channel stability. *Journal of Forestry*, 82 (10), 609-613.

BILOTTA, G.S., BRAZIER, R.E. AND HAYGARTH, P.M., 2007. The impacts of grazing animals on the quality of soils, vegetation, and surface waters in intensively managed grasslands. *Advances in Agronomy*, 94, 237-280.

BINET, F., HALLAIRE, V. AND CURMI, P., 1997. Agricultural practices and the spatial distribution of earthworms in maize fields. Relationships between earthworm abundance, maize plants and soil compaction. *Soil Biology and Biochemistry*, 29 (3-4), 577-583.

BIO INTELLIGENCE SERVICE AND HYDROLOGIC, 2014. *Soil and water in a changing environment.* Final report. Brussels: European Commission.

BIRD, E.C.F., HANSOM, J.D. and MAY, V.J., 2000. Saltmarshes. In: Hansom, J.D. and May, V.J. (Eds.) Coastal geomorphology of Great Britain. Peterborough: Joint Nature Conservation Committee. 515-562.

BIRD, E.C.F., 2008. *Coastal Geomorphology: An Introduction*, 2nd edition, Chichester: Wiley.

BIRKINSHAW, S.J., BATHURST, J.C., IROUMÉ, A. AND PALACIOS, H., 2011. The effect of forest cover on peak flow and sediment discharge – an integrated field and modelling study in central-southern Chile. *Hydrological Processes*, 25 (8), 1284-1297.

BIRKINSHAW, S.J., BATHURST, J.C. AND ROBINSON, M., 2014. 45 years of nonstationary hydrology over a forest plantation growth cycle, Coalburn catchment, Northern England. *Journal of Hydrology*, 519, 559-573.

BLACKWELL, M.S.A. AND MALTBY, E. (ED.), 2006. *Ecoflood guidelines: how to use floodplains for flood risk reduction*. Report for Directorate-General for Research. Luxembourg: Office for Official Publications of the European Communities.

BOARDMAN, J., 1991. Land-use, rainfall and erosion risk on the South Downs. *Soil Use Management*, 7 (1), 34-38.

BOARDMAN, J., 1995. Damage to property by runoff from agricultural land, South Downs, southern England, 1976-93. *The Geographical Journal*, 161 (2), 177-191.

BONN, A., ALLOTT, T., HUBACEK, K. AND STEWART, J., 2009. Managing change in the uplands - challenges in shaping the future. In *Drivers of Environmental Change in Uplands* (ed. A. Bonn, T. Allott, K. Hubacek and J. Stewart), pp. 475-494. Routledge, London and New York.

BORIN, M., PASSONI, M., THIENE, M. AND TEMPESTA, T., 2010. Multiple functions of buffer strips in farming areas. *European Journal of Agronomy*, 32 (1), 103-111.

BOSCH, J.M. AND HEWLETT, J.D., 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. *Journal of Hydrology*, 55 (1–4), 3-23.

BRACKEN, L.J. AND CROKE, J., 2007. The concept of hydrological connectivity and its contribution to understanding runoff-dominated geomorphic systems. *Hydrological Processes*, 21 (13), 1749-1763.

BRADY, N.C., 1974. *The Nature and Properties of Soils*, Eighth Edition, Macmilland Publishing, New York.

BRAZIER, R., PUTTOCK, A., GRAHAM, H., ANDERSON, K., CUNCLIFFE, A. AND ELLIOTT, M., 2016. Quantifying the multiple, environmental benefits of reintroducing the Eurasian beaver. *Geophysical Research Abstracts*, Volume 18, abstract EGU2016-7243.

BRIDGES, T., HENN, R., KOMLOS, S., SCERNO, D., WAMSLEY, T. AND WHITE, K., 2013. *Coastal risk reduction and resilience: using the full array of measures*. CWTS 2013-3. Washington DC: US Army Corps of Engineers, Directorate of Civil Works.

BRIDGES, T., WAGNER, P.W., BURKS-COPES, K.A., BATES, M.E., COLLIER, Z.A., FISCHNICH, C.J., GAILANI, J.Z., LEUCK, L.D., PIERCY, C.D., ROSATI, J.D., RUSSO, E.J., SHAFER, D.J., SUEDEL, B. C., VUXTON, E.A. AND WAMSLEY, T.V., 2015. Use of natural and nature-based features (NNBF) for coastal resilience. North Atlantic Coast comprehensive study: resilient adaptation to increasing risk. Vicksburg, MS: US Army Corps of Engineers, Engineer Research and Development Center.

BRITISH HYDROLOGICAL SOCIETY, 2016. *Circulation – the newsletter of the British Hydrological Society* No 130. Available from: http://www.hydrology.org.uk/assets/Circ%20130_web.pdf [Accessed 4 July 2017].

BROADMEADOW, S. AND NISBET, T.R., 2004. The effects of riparian forest management on the freshwater environment: a literature review of best management practice. *Hydrology and Earth System Sciences*, 8, 286-305.

BROADMEADOW, S.B.; THOMAS, H., AND NISBET, T.R., 2014. Opportunity mapping for woodland creation to reduce diffuse water pollution and flood risk in England and Wales. Farnham, Surrey: Forest Research.

BRONSTERT, A., NIEHOFF, D., AND, BURGER, G., 2002. Effects of climate and land-use change on storm runoff generation: present knowledge and modelling capabilities. *Hydrological Processes*, 16 (2), 509-529.

BRONSTERT, A., AND KUNDZEWICZ, Z. W., 2006. Discussion of the article: CALDER, I. R., AND AYLWARD, B., 2006. Forest and floods: Moving to an evidencebased approach to watershed and integrated flood management. Water International, 31(1) 87-99. *Water International* 31(3): 427–31.

BROOKES, A., GREGORY, K. and DAWSON, F., 1983. An assessment of river channelization in England and Wales. *Science of the Total Environment*, 27, 97-111.

BROOKES, A., 1988. *Channelised Rivers: Perspectives for Environmental Management.* Chichester: Wiley.

BROOKES, A. and SHIELDS, F.D., 1996. *River Channel Restoration: Guiding Principles for Sustainability*. Chichester: John Wiley and Sons.

BROOKS, A.P., 2006. *Design guideline for the reintroduction of wood into Australian streams*. Canberra: Land & Water Australia.

BROUWER, R., KUIK, O., SHEREMET, O., JIANG, Y., BRANDS, D., GERDES, H., LAGO, M., HINZMANN, M., ANGELOPOULOS, N., COWX, I., NICHERSU, I., REICHERT, P., LOGAR, I., PAILLEX, A., SCHUWIRTH, N., LEHTORANTA, V., AROVIITA, J., GARCÍA DE JALÓN, D. and GONZÁLEZ DEL TÁNAGO, M., 2015. Cost-effective restoration measures that promote wider ecosystem and societal benefits. REFORM.

BROWN, K., 2013. *Quantifying bottomland hardwood forest and agricultural grassland evapotranspiration in floodplain reaches of a mid-Missouri stream*. Master's thesis, University of Missouri-Columbia.

BSI, 2005. BS EN 14614:2004. Water quality. Guidance standard for assessing the hydromorphological features of rivers. London: British Standards Institution.

BUCKLER, M., PROCTOR, S., WALKER, J.S., WITTRAM, B., STRATON, P. AND MASKILL, R.M., 2013. *Moors for the Future Partnerships restoration methods for restoring bare peat in the South Pennines SAC: evidence-based recommendations.* Edale, Derbyshire: Moors for the Future Partnership. Available from: <u>http://www.moorsforthefuture.org.uk/research-publications</u> [Accessed 7 July 2017]

BUFFINGTON, J.M. and MONTGOMERY, D.R. 1999. Effects of hydraulic roughness on surface textures of gravel-bed rivers. *Water Resources Research*, 35 (11), 3507-3521.

BUFFINGTON, J.M., LISLE, T.E., WOODSMITH, R.D. AND HILTON, S. 2002. Controls on the size and occurrence of pools in coarse-grained forest rivers. *River Research and Applications*, 18 (6), 507-531.

BUIJSE, A.D., COOPS, H., STARAS, M., JANS, L., VAN GEEST, G., GRIFT, R., IBELINGS, B.W., OOSTERBERG, W. AND ROOZEN, F.C., 2002. Restoration strategies for river floodplains along large lowland rivers in Europe. *Freshwater Biology*, 47 (4), 889-907.

BULLEN CONSULTANTS, 1997. *Lower Witham strategy study.* Report for the Environment Agency.

BULLOCK, F.G., 1992. Crop rotation. *Critical Review of Plant Sciences*, 11 (4), 309-326.

BULLOCK, A. AND ACREMAN, M., 2003. The role of wetlands in the hydrological cycle. *Hydrology and Earth System Sciences*, 7 (3), 358-389.

BULYGINA, N., MCINTYRE, N. AND WHEATER, H., 2009. Conditioning rainfall-runoff model parameters for ungauged catchments and land management impacts analysis. *Hydrology and Earth System Sciences*, 13 (6), 893-904.

BURGESS, C.P., CHAPMAN, R., SINGLETON, O.L. AND THOM, E.R., 2000. Shallow mechanical loosening of a soil under dairy cattle grazing: effects on soil and pasture. *New Zealand Journal of Agricultural Research*, 43 (2), 279-290.

BUTLER, D.R. AND MALANSON, G.P., 1995. Sedimentation rates and patterns in beaver ponds in a mountain environment. *Geomorphology*, 13 (1-4), 255-269.

BUTLER, D.R. and MALANSON, G.P. 2005. The geomorphic influences of beaver dams and failures of beaver dams. *Geomorphology*, 71 (1-2), 48-60.

CALDER, I.R., 1990. Evaporation in the Uplands. Chichester: Wiley.

CALDER, I.R., REID, I., NISBET, T.R., ARMSTRONG, A., GREEN, J.C. AND PARKIN, G., 2002. *Study of the potential impacts on water resources of proposed afforestation.* Report by Loughborough University for the Trees and Drought Project on Lowland England – TaDPoLE. London: Department for Environment, Food and Rural Affairs.

CALDER, I.R., REID, I., NISBET, T.R. AND GREEN, J.C., 2003. Assessing the water use of short vegetation and forests: application of the Hydrological Land Use Change (HYLUC) model. *Water Resources Research*, 39 (11), 1319-1328.

CALDER, I.R., NISBET, T.R., AND HARRISON, J. A., 2009. An evaluation of the impacts of energy tree plantations on water resources in the UK under present and future UKCIP02 climate scenarios. *Water Resources Research*, 45.

CAMPBELL-PALMER, R., GOW, D., CAMPBELL, R., DICKINSON, H., GIRLING, S., FURNELL, J., HALLEY, D., HONES, S., LISLE, S., PARKER, H., SCHWAB, G. AND ROSELL, F., 2016. *The Eurasian Beaver Handbook: Ecology and Management of* Castor fiber. Exeter: Pelagic Publishing.

CANNELL, R.Q., DAVIES, D.B., MACKNEY, D. AND PIDGEON, J.D., 1978. The suitability of soils for sequential direct drilling of combine-harvested crops in Britain: a provisional classification. *Outlook on Agriculture*, 9 (6), 306-316.

CAREY, P.D., WALLIS, S., CHAMBERLAIN, P.M., COOPER, A., EMMETT, B.A., MASKELL, L.C., MCCANN, T., MURPHY, J., NORTON, L.R., REYNOLDS, B., SCOTT, W.A., SIMPSON, I.C., SMART, S.M. AND ULLYETT, J.M., 2008. *Countryside Survey: UK results from 2007.* Wallingford: Centre for Ecology and Hydrology.

CARROLL, Z.L., BIRD, S.B., EMMETT, B.A., REYNOLDS, B. AND SINCLAIR, F.L., 2004a. Investigating the impact of tree shelterbelts on agricultural soils. In *Landscape Ecology of Trees and Forests*, Proceedings of 12th Annual IALE (UK) Conference (Cirencester, 2004), edited by R. Smithers. International Association for Landscape Ecology (UK).

CARROLL, Z.L., REYNOLDS, B., EMMETT, B.A., SINCLAIR, F.L., RUIZ DE ONA, C. AND WILLIAMS, P., 2004b. *The effect of stocking density on soil in upland Wales.* Countryside Council for Wales Science Report Number 630. Bangor: Centre for Ecology and Hydrology.

CASHMAN, M.J., PILOTTO, F., HARVEY, G.L., WHARTON, G. and PUSCH, M.T., 2016. Combined stable-isotope and fatty-acid analyses demonstrate that large wood increases the autochthonous trophic base of a macroinvertebrate assemblage. *Freshwater Biology*, 61 (4), 549-564.

CASTLE, D.A., MCCUNNALL, J. AND TRING, I.M., 1984. *Field Drainage: Principles and Practices*. London: Batsford Academic and Educational.

CHADWICK, D.R. AND CHEN, S., 2002. Manures. In *Agriculture, Hydrology and Water Quality* (ed. P.M. Haygarth and S.C. Jarvis), pp. 437-451. Wallingford: CAB International.

CHAMBERS, B.J., DAVIES, D.B. AND HOLMES, S., 1992. Monitoring of water erosion on arable farms in England and Wales, 1989–90. *Soil Use and Management*, 8 (4), 163-169.

CHANDLER, K.R. AND CHAPPELL, N.A., 2008. Influence of individual oak (*Quercus robur*) trees on saturated hydraulic conductivity. *Forest Ecology and Management*, 256 (5), 1222-1229.

CHARLESWORTH, S., HARKER, E. AND RICKARD, S., 2003. A review of sustainable drainage systems (SuDS): a soft option for hard drainage questions? *Geography*, 88 (2), 99-107.

CHERRY, J. AND BESCHTA, R.L., 1989. Coarse woody debris and channel morphology – a flume study. *Water Resources Bulletin*, 25 (5), 1031-1036.

CHIN, A., LAURENCIO, L.R., DANIELS, M.D., WOHL, E., URBAN, M.A., BOYER, K.L., BUTT, A., PIÉGAY, H. AND GREGORY, K.J., 2014. The significance of perceptions and feedbacks for effectively managing wood in rivers. *River Research and Applications*, 30 (1), 98-111.

CHMURA, D.G.L., ANISFELD, S.C., CAHOON, D.R., LYNCH, J.C., 2003. Global carbon sequestration in tidal, saline wetland soils. *Global Biogeochemical Cycles*, 17 (4), 1111.

CHOW, V.T., 1959. Open Channel Hydraulics. New York: McGraw-Hill.

CHRISTIAN, D.G. AND BALL, B.C., 1994. Reduced cultivation and direct drilling for cereals in Great Britain. In *Conservation Tillage in Temperate Agroecosystems* (ed. M.R. Carter), pp. 117-140. Boca Raton, FL: CRC Press.

CHURCH, M., 2006. Bed material transport and the morphology of alluvial river channels. *Annual Review of Earth and Planetary Sciences*, 34, 325-354.

CIRIA, 2004. Coastal and estuarine managed realignment – design issues. CIRIA Report C628. London: CIRIA.

CIRIA, 2010, *Beach management manual*, 2nd edition. CIRIA Report C685. London: CIRIA.

CIRIA, 2015. The SuDS manual. CIRIA Report C753. London: CIRIA.

CIWEM, 2014. *Floods and dredging – a reality check* [online]. London: Chartered Institution of Water and Environmental Management. Available from: <u>http://www.ciwem.org/policy/flooding/</u> [Accessed 11 July 2017].

CLAESON, S. and COFFIN, B., 2015. Physical and Biological Responses to an Alternative Removal Strategy of a Moderate-sized Dam in Washington, USA. *River Research and Applications*.

CLIMO, W.J. AND RICHARDSON, M.A., 1984. Factors affecting the susceptibility of 3 soils in the Manawatu to stock treading. *New Zealand Journal of Agricultural Research*, 27 (2), 247-253.

CLILVERD, H.M., THOMPSON, J.R., HEPPELL, C.M., SAYER, C.D. and AXMACHER, J.C., 2013. River–floodplain hydrology of an embanked lowland Chalk river and initial response to embankment removal. *Hydrological Sciences Journal-Journal Des Sciences Hydrologiques*, 58 (3), 627-650.

CLILVERD, H., THOMPSON, J., HEPPELL, K., SAYER, C. AND AXMACHER, J., 2015. *Removal of river embankments and the modelled effects on river–floodplain hydrodynamics.* EGU General Assembly Conference Abstracts 2015, abstract 12974.

CLILVERD, H.M., THOMPSON, J.R., HEPPELL, C.M., SAYER, C.D. and AXMACHER, J.C., 2016. Coupled hydrological/hydraulic modelling of river restoration impacts and floodplain hydrodynamics. *River Research and Applications*, 32 (9), 1927-1948.
COGNARD-PLANCQ, A.-L., MARC, V., DIDON-LESCOT, J.-F. AND NORMAND, M., 2001. The role of forest cover on streamflow down sub-Mediterranean mountain watersheds: a modelling approach. *Journal of Hydrology*, 254 (1), 229-243.

COLLINS, A.L. AND WALLING, D.E., 2007. Sources of fine sediment recovered from the channel bed of lowland groundwater-fed catchments in the UK. *Geomorphology*, 88, 120-138.

COLLINS, A.L., XHANG, Y., MCCHESNEY, D., WALLING, D.E., HALEY, S.M. AND SMITH, P., 2012. Sediment source tracing in a lowland agricultural catchment in southern England using a modified procedure combining statistical analysis and numerical modelling. *Science of the Total Environment*, 414 (2012), 301-317.

COOPER, R., HAMA-AZIZ, Z., HISCOCK, K.,. LOVETT, A., DUGDALE, S., SÜNNENBERG, G., NOBLE, L., BEAMISH, J. AND HOVESEN, P., 2017. Assessing the farm-scale impacts of cover crops and non-inversion tillage regimes on nutrient losses from an arable catchment. *Agriculture, Ecosystems and Environment*, 237, 181-193.

CORENBLIT, D., TABACCHI, E., STEIGER, J. AND GURNELL, A.M., 2007. Reciprocal interactions and adjustments between fluvial landforms and vegetation dynamics in river corridors: a review of complementary approaches. *Earth Sciences Review*, 84 (1–2), 56-86.

CORNISH, P.M., 1993. The effects of logging and forest regeneration on water yields in a moist eucalypt forest in New South Wales, Australia. *Journal of Hydrology*, 150 (2-4), 301-322.

COSTIGAN, K., RUFFING, C., PERKIN, J. and DANIELS, M., 2014. Rapid Response of a Sand-Dominated River to Installation and Removal of a Temporary Run-of-the-River Dam. *River Research and Applications*.

COULSON, J. C., BUTTERFIELD, J. E. L. AND HENDERSON, E., 1990. The effect of open drainage ditches on the plant and invertebrate communities of moorland and on the decomposition of peat. *Journal of Applied Ecology*, 549-561.

COWAP, C., WARREN, S., PUTTOCK, A., BRAZIER, R. AND ELLIOT, M., 2016. *The economic value of ecosystems services provided by Culm grasslands*. Exeter: Devon Wildlife trust.

CRONIN, B., 2016. *How soft engineering kept the Yorkshire town of Pickering flood free the winter storms* [online]. New Civil Engineer. Available from: <u>https://www.newcivilengineer.com/technical-excellence/keeping-pickering-flood-free/10003242.article</u> [Accessed 4 July 2017].

CRUE, 2008. *Efficiency of non-structural flood mitigation measures: 'room for the river' and 'retaining water in the landscape'.* CRUE Research Report No. 1-6. Results from the 1st ERA-Net CRUE Funding Imitative. London: CRUE. Available from: <u>http://www.crue-eranet.net</u> [Accessed 4 July 2017].

CURRAN, J.H. and WOHL, E.E., 2003. Large woody debris and flow resistance in step-pool channels, Cascade Range, Washington. *Geomorphology*, 51 (1-3), 141-157.

CURRAN COURNANE, F., MCDOWELL, R.W., LITTLEJOHN, R.P., HOULBROOKE, D.J. AND CONDRON, L.M., 2011. Is mechanical soil aeration a strategy to alleviate soil compaction and decrease phosphorus and suspended sediment losses from irrigated and rain-fed cattle-grazed pastures? *Soil Use and Management*, 27 (3), 376-384.

DARBY, S.E. and SIMON, A., 1999. *Incised River Channels: Processes Forms Engineering and Management*. Chichester: Wiley.

DA SILVA, L.V., 2012. *Ecosystem services assessment at Steart Peninsula, Somerset, UK*. Master's thesis, Imperial College London.

DA SILVA, L.V., EVARARD, M. AND SHORE, R.G., 2014. Ecosystem services assessment at Steart Peninsula, Somerset, UK. *Ecosystem Services*, 10, 19-34.

DAVIES, B., EAGLE, D. AND FINNEY, B., 1972. *Soil Management*. Ipswich: Farming Press.

DEASY, C., QUINTON, J.N., SILGRAM, M., BAILEY, A.P., JACKSON, B. AND STEVENS, C.J., 2009. Mitigation options for sediment and phosphorus loss from winter-sown arable crops. *Journal of environmental Quality*, 38 (5), 2121-2130.

DEASY, C., QUINTON, J.N., SILGRAM, M., STOATE, C., JACKSON, R., STEVENS, C.J. AND BAILEY, A.P., 2010. Mitigation Options for Phosphorus and Sediment (MOPS): reducing pollution in run-off from arable fields. *The Environmentalist*, Issue 8, 12-17.

DEASY, C., TITMAN, A. AND QUINTON, J., 2014. Measurement of flood peak effects as a result of soil and land management, with focus on experimental issues and scale. *Journal of Environmental Management*, 132, 304-312.

DEFRA, 2004a. *Making Space for Water. Developing a new government strategy for flood and coastal erosion risk management in England. A consultation exercise.* London: Department for Environment, Food and Rural Affairs.

DEFRA, 2004b. *Rural Strategy 2004*. London: Department for Environment, Food and Rural Affairs.

DEFRA, 2005. *Making Space for Water. Taking forward a new government strategy for flood and coastal risk management in England. First response.* London: Department for Environment, Food and Rural Affairs.

DEFRA, 2006. Shoreline management plan guidance. Volume 1: Aims and requirements. London: Department for Environment, Food and Rural Affairs.

DEFRA, 2009. *Protecting our water, soil and air: a code of good agricultural practice for farmers, growers and land managers*. London: Department for Environment, Food and Rural Affairs.

DEFRA, 2016. *Key results by UK country: 1866 to 2016* [online]. Statistical data set: Structure of the agricultural industry in England and the UK at June. London: Defra. Available from: <u>https://www.gov.uk/government/statistical-data-sets/structure-of-the-agricultural-industry-in-england-and-the-uk-at-june</u> [Accessed 10 May 2017].

DEFRA, 2017. *Crop rotation* [online]. In *Integrated Crop Management (CPA Book)*, published by the Crop Protection Association in 1996 and made available online by Defra. Available from:

http://adlib.everysite.co.uk/adlib/defra/content.aspx?id=000IL3890W.17USY7NEWZ4R 1 [Accessed 4 July 2017].

DE JONG, B., KEIJSERS, J.G.S., RIKSEN, M.J.P.M., KROL, J. AND SLIM, P.A., 2014. Soft engineering vs. a dynamic approach in coastal dune management: a case study on the North Sea barrier island of Ameland, the Netherlands. *Journal of Coastal Research*, 30 (4), 670-684.

DELANEY, T.A., 1995. Benefits to downstream flood attenuation and water quality as a result of constructed wetlands in agricultural landscapes. *Journal of Soil and Water Conservation*, 50 (6), 620-626.

DELTARES, 2013. *Eco-engineering in the Netherlands. Soft interventions with a solid impact.* The Hague: Ministry of Infrastructure and the Environment.

DE VISSCHER, M., NYSSEN, J., PONTZEELE, J., BILLI, P. AND FRANKL, A., 2014. Spatio-temporal sedimentation patterns in beaver ponds along the Chevral river, Ardennes, Belgium. *Hydrological Processes*, 28 (4), 1602-1615.

DEWALD, C.L., HENRY, J., BRUCKERHOFF, S., RITCHIE, J., DABNEY, S., SHEPHERD, D. AND WOLF, D., 1996. Guidelines for establishing warm season grass hedges for erosion control. *Journal of Soil and Water Conservation*, 51 (1), 16-20.

DHI, 2009. MIKE SHE user documentation. Hørsholm, Denmark: DHI.

DICK, R.P., 1992. A review: long-term effects of agricultural systems on soil biochemical and microbial parameters. *Agriculture, Ecosystems and Environment*, 40 (1-4), 25-36.

DILLAHA, T.A., SHERRARD, J.H. AND LEE, D., 1986. *Long-term effectiveness and maintenance of vegetative filter strips*. Bulletin 153. Blacksberg, VA: Virginia Water Resources Research Center, Virginia Polytechnic Institute and State University.

DIXON, S.J., 2013. *Investigating the effects of large wood and forest management on flood risk and flood hydrology.* PhD thesis, University of Southampton.

DIXON, G. AND SCOTT, M., 2017. *Flood management and forestry at Southwell.* Final JBA contract report to Forestry Commission. Saltaire, Shipley: JBA Consulting.

DIXON, S.J. AND SEAR, D.A., 2014. The influence of geomorphology on large wood dynamics in a low gradient headwater stream. *Water Resources Research*, *50*(12), pp.9194-9210.

DIXON, S.J., SEAR, D.A., ODONI, N.A., SYKES T. AND LANE S.N., 2016. The effects of river restoration on catchment scale flood risk and flood hydrology. *Earth Surface Processes and Landforms*, 41 (7), 997-1008.

DKKV, 2004. *Hochwasservorsorge in Deutschland. Lernen aus der Katastrophe 2002 im Elbegebiet.* Schriftenreihe des DKKV, No. 29. Bonn: German Committee for Disaster Risk Management.

DODD, J., NEWTON, M. AND ADAMS, C. 2016. *The effect of natural flood management in-stream wood placements on fish movement in Scotland*. CD2015_02. Aberdeen: Centre for Expertise in Water (CREW).

DOUGLAS, J.T., KOPPI, A.J., AND CRAWFORD, C.E., 1998. Structural improvement in a grassland soil after changes to wheel-traffic systems to avoid soil compaction. *Soil Use and Management*, 14 (1), 14-18.

DOWNS, P.W. and THORNE, C.R., 2000. Rehabilitation of a lowland river: reconciling flood defence with habitat diversity and geomorphological sustainability. *Journal of Environmental Management*, 58 (4), 249-268.

DOWNS, P.W. and KONDOLF, G.M., 2002. Post-project appraisals in adaptive management of river channel restoration. *Environmental Management*, 29 (4), 477-496.

DREWRY, J.J., LOWE, J.A.H. AND PATON, R.J., 2000. Effect of subsoiling on soil physical properties and pasture production on a Pallic soil in Southland, New Zealand. *New Zealand Journal of Agricultural Research*, 43 (2), 269–277.

DRIVER, A., 2016. *Multiple benefits of river and wetland restoration – measured from projects implemented on the ground*. Unpublished report, Environment Agency, Britsol.

DUFOUR, S. and PIÉGAY, H. 2009. From the myth of a lost paradise to targeted river restoration: forget natural references and focus on human benefits. *River Research and Applications*, 25 (5), 568-581.

DUFFY, A., MOIR, S, BERWICK, N., SHABASHOW, J., D'ARCY, B. AND WADE, R., 2016. *CREW Rural sustainable drainage systems. A practical design and build guide for Scotland's farmers and landowners* CRW2015/2.2. Aberdeen: Centre for Expertise in Water (CREW).

DUNNE, T. AND BLACK, R.D., 1970. Partial area contribution to storm runoff in a small New England catchment. *Water Resources Research*, 6 (5), 1298-1311.

EDEN DTC, 2017. A National Demonstration Test Catchment [online]. Available from: http://www.edendtc.org.uk [Accessed 4 July 2017]

EDWARDS, A. C. AND WITHERS, P. J. A., 2008. Transport and delivery of suspended solids, nitrogen and phosphorus from various sources to freshwaters in the UK. *Journal of Hydrology*, 350(3), 144-153.

EEA, 2015. *Water-retention potential of Europe's forests. A European overview to support natural water-retention measures.* Technical Report No.13/2015. Copenhagen: European Environment Agency.

EEA, 2016. Flood risks and environmental vulnerability: Exploring the synergies between floodplain restoration, water policies and thematic policies. EEA Report No 1/2016. Copenhagen: European Environment Agency.

EEKHOUT, J., FRAAIJE, R. and HOITINK, A., 2014. Morphodynamic regime change in a reconstructed lowland stream. *Earth Surface Dynamics*, 2, 279-293.

EFTEC, 2007. Flood and coastal erosion risk management: economic valuation of environmental effects handbook. London: Eftec.

EFTEC, 2010. Flood and coastal erosion risk management: economic valuation of environmental effects. Handbook for the Environment Agency for England and Wales. Revised version. London: Eftec. Available from:

http://www.gov.uk/government/uploads/system/uploads/attachment_data/file/487240/LI T 10352.pdf [Accessed 4 July 2017]

EFTEC, 2015. *Beam Parklands Natural Capital Account: final report.* Report for the Greater London Authority.

ELLIS, J.B., 1992. Quality issues of source control. In *Urban Drainage: The Natural Way*, Proceedings of CONFLO 92: Integrated Catchment Planning and Source Control (Oxford, 1992). Clevedon, Avon: Hydro Research & Development.

EMPSON, B., COLLINS, T., LEAFE and LOWE, J., 1997. Sustainable flood defence and habitat conservation in estuaries – a strategic framework. Proceedings of 32nd MAFF Conference of River and Coastal Engineers 1997; F2.1-F2.12.

EMMETT, B.A, REYNOLDS, B., CHAMBERLAIN, P.M., ROWE, E., SPURGEON, D., BRITTAIN, S.A., FROGBROOK, Z., HUGHES, S., LAWLOR, A.J., POSKITT, J., POTTER, E., ROBINSON, D.A., SCOTT, A., WOOD, C. AND WOODS, C., 2010. *Countryside Survey: Soils report from 2007.* Countryside Survey Technical Report No. 9/07. Wallingford: Centre for Ecology and Hydrology. Available from: http://www.countrysidesurvey.org.uk/content/soils-report-2007 [Accessed 4 July 2017].

ENGLAND, J. AND GURNELL, A.M., 2016. Incorporating catchment to reach scale processes into hydromorphological assessment in the UK. *Water and Environment Journal*, 30 (1-2), 22-30.

ENVIRONMENT AGENCY, 1996. *The restoration of floodplain woodlands in lowland Britain: a scoping study and recommendations for research*, R&D Technical Report W15. Bristol: Environment Agency.

ENVIRONMENT AGENCY, 1999. *Large wood debris in British headwater rivers: summary report.* Technical Report W185. Bristol: Environment Agency,

ENVIRONMENT AGENCY, 2000. *River Calder, Wakefield Flood Alleviation Scheme. National Review Group Report.* Unpublished report.

ENVIRONMENT AGENCY, 2002a. *Erosion control in maize*. Technical Report P2-123/TR. Bristol: Environment Agency.

ENVIRONMENT AGENCY, 2002b. *Impact of agricultural soil conditions on floods – August 2000.* R&D Technical Report W5B-026/TR. Bristol: Environment Agency.

ENVIRONMENT AGENCY, 2003a. *River habitat survey in Britain and Ireland. Field Survey guidance manual: 2003 version.* Bristol: Environment Agency.

ENVIRONMENT AGENCY, 2003b. *Review of soil management techniques for water retention and minimising diffuse water pollution in the River Parrett catchment*. R&D Technical Report P2-261/10/TR. Bristol: Environment Agency.

ENVIRONMENT AGENCY, 2004. *Ribble Estuary Managed Realignment Study Stage 1. Initial review of feasibility of managed realignment at various sites.* Unpublished report.

ENVIRONMENT AGENCY, 2005. Project appraisal report – Humber Estuary tidal defences, Alkborough Tidal Defence Scheme. Newcastle: Environment Agency, North East.

ENVIRONMENT AGENCY, 2007. *R&D update review of the impact of land use and management on flooding*. Bristol: Environment Agency. Available from: <u>http://webarchive.nationalarchives.gov.uk/20130123162956/http://www.defra.gov.uk/environment/flooding/documents/manage/updatereview.pdf</u> [Accessed 4 July 2017].

ENVIRONMENT AGENCY, 2008a. *Think Soils manual. Soil assessment to avoid erosion and runoff* [online]. Bristol: Environment Agency. Available from: http://adlib.everysite.co.uk/adlib/defra/content.aspx?id=263233 [Accessed 4 July 2017].

ENVIRONMENT AGENCY, 2008b. *Wallasea Island Habitat Creation. Project Appraisal Report.* Unpublished report.

ENVIRONMENT AGENCY, 2009. *Ecosystem services case studies. Better regulation science programme*. Bristol: Environment Agency.

ENVIRONMENT AGENCY, 2010a. *The state of river habitats in England, Wales and the Isle of Man: a snapshot.* Bristol: Environment Agency.

ENVIRONMENT AGENCY, 2010b. *Medmerry managed realignment. Project Appraisal Report.* Unpublished report.

ENVIRONMENT AGENCY, 2010c. *Medmerry managed realignment. Environmental Statement.* Unpublished report.

ENVIRONMENT AGENCY, 2012. *Rural sustainable drainage systems manual.* Bristol: Environment Agency.

ENVIRONMENT AGENCY, 2013a. *Humber Estuary managed realignments: lessons for the future.* Unpublished report.

ENVIRONMENT AGENCY, 2013b. *Beach modelling; lessons learnt from past scheme performance*. SC110004/R. Bristol: Environment Agency.

ENVIRONMENT AGENCY, 2014a. *Working with natural processes to reduce flood risk. R&D framework: science report.* Report SC130004/R2. Bristol: Environment Agency.

ENVIRONMENT AGENCY, 2014b. Aquatic and riparian plant management: controls for vegetation in watercourses. Technical guide. Report SC120008/R2. Bristol: Environment Agency.

ENVIRONMENT AGENCY, 2015a. Long term costing tool: summary of evidence on cost estimation. Report SC080039/R1. Bristol: Environment Agency.

ENVIRONMENT AGENCY, 2015b. *Energy crops and floodplain flows*. Report SC060092/R2. Bristol: Environment Agency.

ENVIRONMENT AGENCY, 2015c. Cost estimation for land use and run-off – summary of evidence. Report SC080039/R12. Bristol: Environment Agency.

ENVIRONMENT AGENCY, 2015d. Somerset Levels and Moors: reducing the risk of flooding [online]. Bristol: Environment Agency. Available from: <u>https://www.gov.uk/government/publications/somerset-levels-and-moors-reducing-the-risk-of-flooding/somerset-levels-and-moors-reducing-the-risk-of-flooding</u> [Accessed 4 July 2017].

ENVIRONMENT AGENCY, 2015e. *Channel management handbook*. Report SC110002. Bristol: Environment Agency.

ENVIRONMENT AGENCY, 2015f. Cost estimation for managed realignment – summary of evidence. Report SC080039/R8. Bristol: Environment Agency.

ENVIRONMENT AGENCY, 2016a. *How to model and map catchment processes when flood risk management planning*. Project SC120015/R1. Bristol: Environment Agency.

ENVIRONMENT AGENCY, 2016b. *Rye Harbour Farm habitat creation project.* Unpublished report.

ENVIRONMENT AGENCY, 2016c. *Design, operation and adaptation of reservoirs for flood storage*. SC120001/R. Bristol: Environment Agency.

ENVIRONMENT AGENCY, undated a. *Woody debris in rivers (England and Wales)* – *supplementary case studies*. Unpublished.

ENVIRONMENT AGENCY, undated b. *Beneficial use of dredgings toolkit*. Unpublished.

ENVIRONMENT AGENCY, undated c. *Testing the USACE models on our shores* – *Lincolnshire, Mablethorpe to Skegness ('Lincshore').* Unpublished.

ENVIRONMENT AGENCY, HASKONINGS AND DEFRA, 2007. Saltmarsh management manual. R&D Technical Report SC030220. Bristol: Environment Agency.

ENVIRONMENT AGENCY MIDLANDS REGION 2000. *Melton Mowbray Flood Alleviation Scheme, Project Appraisal Report.* Solihull: Environment Agency.

ERWIN, S.O., SCHMIDT, J.C. AND ALLRED, T.M. 2016. Post-project geomorphic assessment of a large process-based river restoration project. *Geomorphology*, 270, 145-158.

ESSEX WILDLIFE TRUST, 2005. *Abbotts Hall Farm Fact Sheet 9. Lessons learned from realignment.* Colchester: Essex Wildlife Trust.

ESTEVES, L.S., 2014. *Managed realignment: a viable long-term coastal management strategy?* Springer Briefs in Environmental Science. New York: Springer.

EVANS, E.P., ASHELY, R., HALL, J., PENNING-ROWSELL, E., SAYERS, P., THORNE, C. AND WATKINSON, A., 2004. *Foresight. Future Flooding. Scientific Summary. Volume II: Managing future risks.* London: Office of Science and Technology.

EVANS, M., ALLOTT, T., HOLDEN, J., FLITCROFT, C. AND BONN, A., 2005. *Understanding gully blocking in deep peat.* Moors for the Future Report No 4. Castleton, Derbyshire: Moors for the Future. Available from: <u>http://www.moorsforthefuture.org.uk/research-publications</u> [Accessed 7 July 2017].

EVANS, R., 2006. Land use, sediment delivery and sediment yield in England and Wales. In *Soil Erosion and Sediment Redistribution in River Catchments: Measurement, Modelling and Management* (ed. P. Owens and A. Collins), pp. 70-84. Wallingford: CABI Publishing.

EVANS, R. AND BOARDMAN, J., 2003. Curtailment of muddy floods in the Sompting catchment, South Downs, West Sussex, southern England. *Soil Use and Management*, 19 (3), 223-231.

EVERARD, M. and MCINNES, R. 2013. Systemic solutions for multi-benefit water and environmental management. *Science of the Total Environment*, 461-462, 170-179.

EVERARD, O., PERSOONS, E., VANDAELE, K. AND VAN WESEMAEL, B., 2007. Effectiveness of erosion mitigation measures to prevent muddy floods: a case study in the Belgian loam belt. *Agriculture, Ecosystems and Environment*, 118 (1), 149-158.

EVERARD, O., VANDAELE, K., VAN WESEMAEL, B. AND BIELDERS, C.L., 2008. A grassed waterway and earthen dams to control muddy floods from a cultivated catchment of the Belgian loess belt. *Geomorphology*, 100 (3-4), 419-428.

FAHEY, B. AND JACKSON, R., 1997. Hydrological impacts of converting native forests and grasslands to pine plantations, South Island, New Zealand. *Agricultural and Forest Meteorology*, 84 (1), 69-82.

FALLOON, P. AND BETTS, R. 2010. Climate impacts on European agriculture and water management in the context of adaptation and mitigation—the importance of an integrated approach. *Science of the Total Environment*, 408 (23), 5667-5687.

FEDERAL INTERAGENCY STREAM RESTORATION WORKING GROUP, 2001. Stream corridor restoration: principles, processes, and practices. Washington DC: US Department of Agriculture.

FELD, C. K., BIRK, S., BRADLEY, D.C., HERING, D., KAIL, J., MARZIN, A., MELCHER, A., NEMITZ, D., PEDERSON, M.L., PLETTERBAUER, F., PONT, D., VERDONSCHOT, P.F.M., and FRIBERG, N., 2011. From natrual to degraded rivers and back again: a test of restoration ecology theory and practice. *Advances in Ecological Research*, Submitted.

FERRARI, M.R., MILLER, J.R. and RUSSELL, G.L., 1999. Modeling the effect of wetlands, flooding, and irrigation on river flow: application to the Aral Sea. *Water Resources Research*, 35 (6), 1869-1876.

FEUSTEL, I.C. AND BYERS, H.G., 1936. *The comparative moisture-absorbing and moisture-retaining capacities of peat and soil mixtures*. Technical Bulletin no. 532. Washington DC: United States Department of Agriculture.

FIENER, P. AND AUERSWALD, K., 2003. Effectiveness of grassed waterways in reducing runoff and sediment delivery from agricultural watersheds. *Journal of Environmental Quality*, 32 (3), 927-936.

FIENER, P. AND AUERSWALD, K., 2006. Seasonal variation of grassed waterway effectiveness in reducing runoff and sediment delivery from agricultural watersheds in temperate Europe. *Soil and Tillage Research*, 87 (1), 48-58.

FIENER, P. AUERSWALD, K. AND WEIGAND, S., 2005. Managing erosion and water quality in agricultural watersheds by small detention ponds. *Agriculture, Ecosystems and Environment*, 110 (3), 132-142.

FLORES, L., LARRANAGA, A., DIEZ, J. and ELOSEGI, A., 2011. Experimental wood addition in streams: effects on organic matter storage and breakdown. *Freshwater Biology*, 56 (10), 2156-2167.

FLORSHEIM, J., MOUNT, J. and CONSTANTINE, C., 2006. A geomorphic monitoring and adaptive assessment framework to assess the effect of lowland floodplain river restoration on channel–floodplain sediment continuity. *River Research and Applications*, 22, 353-375.

FLOW PARTNERSHIP, 2017. *The Flow Partnership. Rejuvenating desertified and flooded lands with community collaboration* [online]. Available from: https://www.theflowpartnership.org [Accessed 6 July 2017].

FORD, D., DYKE, P. AND WOODHALL, A., 2009. *Case study: South Milton Sands removing sea defences and allowing natural processes*. Devon: National Trust.

FORD, H., GARBUTT, A., LADD, C., MALARKEY, J. AND SKOV, M.W., 2016. Soil stabilization linked to plant diversity and environmental context in coastal wetlands. *Journal of Vegetation Science*, 27 (2), 259-268.

FOREST RESEARCH, 2008. *The Robinwood Robinflood report. Evaluation of large woody debris in watercourses.* Report for the Robinwood Project, an INTEREG IIIc Regional Framework Operation project. Farnham, Surrey: Forest Research.

FORESTRY COMMISSION, 2011. *The UK Forestry Standard*. Edinburgh: Forestry Commission.

FORESTRY COMMISSION, 2014. UK Woodland Carbon Code: requirements for voluntary carbon sequestration projects; Version 1.3. Edinburgh: Forestry Commission.

FOREST ENTERPRISE ENGLAND, 2016. *Natural capital accounts 2015/16*. Bristol: Forestry Commission England.

FOWLER, H.J., 2005. Are extremes increasing? Changing rainfall patterns in Yorkshire. *The Yorkshire and Humberside Regional Review*, 15 (1), 21-24.

FRANCIS, C.A. AND CLEGG, M.D., 1990. Crop rotations in sustainable agricultural systems. In *Sustainable Agriculture Systems* (ed. C.A. Edwards, R. Lal, P. Madden, R.H. Miller and G. House), pp. 107-122. Ankeny, IA: Soil and Water Conservation Society.

FRANKLIN, D.H., CABRERA, M.L., WEST, L.T., CALVERT, V.H. AND REMA, J.A., 2007. Aerating grasslands: effects on runoff and phosphorus loss from applied broiler litter. *Journal of Environmental Quality*, 36 (1), 208–215.

FRIEDRICH, T. AND KASSAM, A., 2012. No-till farming and the environment: do no tillage systems require more chemicals?. *Outlooks on Pest Management*, 23 (4), pp. 153-157.

FULLER, R.J. AND GOUGH, S.G., 1999. Changes in sheep numbers in Britain: implications for bird populations. *Biological Conservation*, 91 (1), 73-89.

FWAG SOUTH WEST, 2016. *Flood management information: cross drains.* Hills to Levels Information Sheet. Tetbury, Gloucestershire: Farming & Wildlife Advisory Group South West.

GAMBRELL, P.R., 1994. Trace and toxic metals in wetlands: a review. *Journal of Environmental Quality*, 23: 883-891.

GAO, J., HOLDEN, J. AND KIRKBY, M., 2015. A distributed TOPMODEL for modelling impacts of land-cover change on river flow in upland peatland catchments. *Hydrological Processes*, 29 (13), 2867-2879.

GAO, J., HOLDEN, J. AND KIRKBY, M., 2016. The impact of land-cover change on flood peaks in peatland basins. *Water Resources Research*, 52 (5), 3477-3492.

GARDESTRÖM, J., HOLMQVIST, D., POLVI, L. E. and NILSSON, C., 2013. Demonstration restoration measures in tributaries of the Vindel River catchment. *Ecology & society*, 18, Article Number: UNSP 8.

GAYWOOD, M., 2015. *Beavers in Scotland: a report to the Scottish Government*. Inverness: Scottish Natural Heritage.

GERIS, J.R.M.C., 2012. *Multiscale impacts of land use/management changes on flood response in the river Hodder catchment, north-west England, Newcastle upon Tyne.* PhD thesis, Newcastle University.

GHAVASIEH, A.R., POULARD, C., ND PASQUIER, A., 2006. Effect of roughened strips on flood propagation: assessment on representative virtual cases and validation. *Journal of Hydrology*, 318 (1), 121-137.

GHIMIRE, S., 2013. Application of a 2D hydrodynamic model for assessing flood risk from extreme storm events. *Climate*, 1 (3), 148-162.

GHIMIRE, S., WILKINSON, M.E. AND DONALDSON-SELBY, G., 2014. Assessing the effectiveness of Natural Flood Management (NFM) in a rural catchment of Scotland. In *Informatics and the Environment: Data and Model Integration in a Heterogeneous Hydro World*, Proceedings of 11th International Conference on Hydroinformatics (New York, 2014), pp. 482-489. Red Hook, NY: Curran Associates.

GIFFORD, G.F. AND HAWKINS, R.H., 1978. Hydrologic impact of grazing on infiltration: a critical review. *Water Resources Research*, 14 (2), 305-313.

GILLIAM, J.W., 1994. Riparian Wetlands and Water Quality. *Journal of Environmental Quality* 23: 896-900.

GILVEAR, D.J., 1999. Fluvial geomorphology and river engineering: future roles utilizing a fluvial hydrosystems framework. *Geomorphology*, 31 (1-4), 229-245.

GILVEAR, D.J. and BLACK, A.R., 1999. Flood-induced embankment failures on the River Tay: implications of climatically induced hydrological change in Scotland. *Hydrological Sciences Journal-Journal Des Sciences Hydrologiques*, 44 (3), 345-362.

GILVEAR, D.J., CASAS-MULET, R. AND SPRAY, C.J., 2012. Trends and issues in delivery of integrated catchment scale river restoration: lessons learned from a national river restoration survey within Scotland. *River Research Applications*, 28, 234-246.

GILVEAR, D. J., SPRAY, C. J. and CASAS-MULET, R., 2013. River rehabilitation for the delivery of multiple ecosystem services at the river network scale. *Journal of environmental management*, 126, 30-43.

GIPPEL, C.J., 1995. Environmental hydraulics of large woody debris in streams and rivers. Journal of Environmental Engineering, 121 (5), 388-395.

GIRIAT, D., GORCZYCA, E. AND SOBUCKI, M., 2016. Beaver ponds' impact on fluvial processes (Beskid Niski Mts., SE Poland). *Science of the Total Environment*, 544, 339-353.

GOECK, J. AND GEISELER, G., 1989. Erosion control in maize fields in Schleswif-Holstein (FRG). In: *Soil erosion protection measures in Europe* (eds SCHWERTMANN, U., RICKSON, R.J. AND AUERSWALK, K.), Soil Technology Series I, pp.83-92.

GOODELL, B.C., 1959. Management of forest stands in western United States to influence the flow of snow-fed streams. In *Proceedings of Symposium of Hannoversch-Munden, Volume 1: Water and Woodlands,* Publication 48, pp. 49-58. Gentbrugge, Belgium: International Association of Scientific Hydrology.

GOWING, D., LAWSON, C., YOUNGS, E., BARBER, K., RODWELL, J., PROSSER, M., WALLACE, H., MOUNTFORD, J. AND SPOOR, G., 2002. *The water regime requirements and the response to hydrological change of grassland plant communities.* Final report to Defra (Conservation Management Division), Project BD1310.

GRABLE, A.R., 1966. Soil aeration and plant growth. *Advances in Agronomy*, 18, 57-106.

GRAYSON, R., HOLDEN, J. AND ROSE, R., 2010. Long-term change in storm hydrographs in response to peatland vegetation change. *Journal of Hydrology*, 389 (3-4), 336-343.

GREAT FEN TEAM, undated. *Managing dykes, ditches and drains. Water: to drain or retain?* [online]. Ramsay Heights, Cambridgeshire: Great Fen Team. Available from: <u>http://www.greatfen.org.uk/restoration/dykes-ditches-and-drains</u> [Accessed 6 July 2017].

GREEN, T.R., AHUJA, L.R., AND BENJAMIN, J.G., 2003. Advances and challenges in predicting agricultural management effects on soil hydraulic properties. *Geoderma*, 116 (1), 3-27.

GREEN, K.C. AND WESTBROOK, C.J., 2009. Changes in riparian area structure, channel hydraulics, and sediment yield following loss of beaver dams. *BC Journal of Ecosystems and Management*, 10 (1), 68-79.

GREENWOOD, K.L. AND MCKENZIE, B.M., 2001. Grazing effects on soil physical properties and the consequences for pastures: a review. *Australian Journal of Experimental Agriculture*, 41 (8), 1231-1250.

GREENWOOD, K.L., MACLEOD, D.A. AND HUTHCHINSON, K.J., 1997. Long-term stocking rate effects on soil physical properties. *Australian Journal of Exper-imental Agriculture* 37: 413–419.

GREGORY, K.J., GURNELL, A.M. AND HILL, C.T., 1985. The permanence of debris dams related to river channel processes. *Hydrological Sciences Journal-Journal Des Sciences Hydrologiques*, 30 (3), 371-381.

GREGORY, K.J., DAVIS, R.J. AND TOOTH, S., 1993. Spatial-distribution of coarse woody debris dams in the Lymington Basin, Hampshire, UK. *Geomorphology*, 6 (3), 207-224.

GREGORY, K.J., GURNELL, A.M., HILL, C.T. AND TOOTH, S.,1994. Stability of the pool riffle sequence in changing river channels. *Regulated Rivers: Research and Management*, 9 (1), 35-43.

GRIFFIN, I., PERFECT, C. AND WALLACE, M., 2015. *River restoration and biodiversity*. Scottish Natural Heritage Commissioned Report No. 817. Inverness: Scottish Natural Heritage.

GRYGORUK, M. AND NOWAK, M., 2014. Spatial and temporal variability of channel retention in a lowland temperate forest stream settled by European beaver (*Castor fiber*). *Forests*, 5, 2276-2288.

GRYGORUK, M., MIROSŁAW-ŚWIĄTEK, D., CHRZANOWSKA, W. AND IGNAR, S., 2013. How much for water? Economic assessment and mapping of floodplain water storage as a catchment-scale ecosystem service of wetlands. *Water*, 5 (4), 1760-1779.

GUERRIN, J., 2015. A floodplain restoration project on the River Rhône (France): analyzing challenges to its implementation. *Regional Environmental Change*, 15 (3), 559-568.

GUILLEMETTE, F., PLAMONDON, A.P., PREVOST, M. AND LEVESQUE, D., 2005. Rainfall generated stormflow response to clearcutting a boreal forest: peak flow comparison with 50 world-wide basin studies. *Journal of Hydrology*, 302 (1-4), 137– 153.

GUMIERE, S. J., LE BISSONNAIS, Y., RACLOT, D. AND CHEVIRON, B., 2011. Vegetated filter effects on sedimentological connectivity of agricultural catchments in erosion modelling: a review. *Earth Surface Processes and Landforms*, 36(1), 3-19.

GURNELL, A.M., 1998. The hydrogeomorphological effects of beaver dam-building activity. *Progress in Physical Geography*, 22 (2), 167.

GURNELL, A.M., 2013. Wood in fluvial systems. In *Treatise on Geomorphology* (ed. J.F. Shroder), pp. 163-188. San Diego: Academic Press.

GURNELL, A., 2014. Plants as river system engineers. *Earth Surface Processes and Landforms*, 39 (1), 4-25.

GURNELL, A.M. and SWEET, R., 1998. The distribution of large woody debris accumulations and pools in relation to woodland stream management in a small, low-gradient stream. *Earth Surface Processes and Landforms*, 23 (12), 1101-1121.

GURNELL, A.M. AND GRABOWSKI, R.C., 2016. Vegetation-hydrogeomorphology interactions in a low-energy, human-impacted river. *River Research and Applications*, 32 (2), 202-215.

GURNELL, J., GURNELL, A.M., DEMERITT, D., LURZ, P.W.W., SHIRLEY, M.D.F., RUSHTON, S.P., FAULKES, C.G., NOBERT, S. and HARE, E.J., 2009. *The feasibility and acceptability of reintroducing the European beaver to England*. Natural England Commissioned Report NECR002. Report commissioned jointly by Natural England and the People's Trust For Endangered Species. Worcester: Natural England

GURNELL, A.M., BERTOLDI, W., TOCKNER, K., WHARTON, G. AND ZOLEZZI, G., 2016. How large is a river? Conceptualizing river landscape signatures and envelopes in four dimensions. *Wiley Interdisciplinary Reviews: Water* 3 (3), 313-325.

GUSTARD, A., ROALD, L.A., DEMUTH, S., LUMADJENG, H.S. AND GROSS, R., 1989. *Flow regimes from experimental and network data (FREND) I and II*. Wallingfrod: Institute of Hydrology.

HABERSACK, H., HAUER, C., SCHOBER, B., DISTER, E., QUICK, I., HARMS, O., DOPKE, M., WINTZ, M. AND PIQUETTE, E., 2008. *Flood risk reduction by PReserving and restOring river Floodplains (PRO_Floodplain)*. CRUE Research Report No I-3. Results from the 1st ERA-Net CRUE Funding Imitative. London: CRUE. Available from: http://www.crue-eranet.net [Accessed 4 July 2017].

HALCROW, 2012. Coastal processes report. Unpublished report.

HANSEN, B., SCHØNNING, P. AND SIBBESEN, E., 1999. Roughness indices for estimation of depression storage capacity of tilled soil surfaces. *Soil and Tillage Research*, 52 (1-2), 103-111.

HAMMERSMARK, C.T., RAINS, M.C. AND MOUNT, J.F., 2008. Quantifying the hydrological effects of stream restoration in a montane meadow, northern California, USA. *River Research and Applications*, 24 (6), 735-753.

HARDIMAN, N., CUNNINGHAM, R. AND JONSTONOVA, A., 2009. *Technical review of NFRM techniques, their effectiveness and wider benefits.* Sandy, Bedfordshire: RSPB.

HARDING, R.J., HALL, R.L., NEAL, C., ROBERTS, J.M., ROSIER, P.T.W. AND KINNIBURGH, D.G., 1992. *Hydrological impacts of broadleaf woodlands: implications for water use and water quality.* Project Report 115/03/ST. Bristol: National Rivers Authority.

HARDY, R.J., BATES, P.D. AND ANDERSON, M.G., 2000. Modelling suspended sediment deposition on a fluvial floodplain using a two-dimensional dynamic finite element model. *Journal of Hydrology*, 229 (3-4), 202-218.

HARMON, M.E., FRANKLIN, J.F., SWANSON, F.J., SOLLINS, P., GREGORY, S.V., LATTIN, J.D., ANDERSON, N.H., CLINE, S.P., AUMEN, N.G., SEDELL, J.R., LIENKAEMPER, G.W., CROMACK, K. AND CUMMINS, K.W., 1986. Ecology of coarse woody debris in temperate ecosystems. *Advances in Ecological Research*, 15, 133-302.

HARR, R.D., 1986. Effects of clearcutting on rain-on-snow runoff in western Oregon: a new look at old studies. *Water Resources Research*, 22 (7), 1095-1100.

HARRIS, G.L., CLEMENTS, R.O., ROSE, S.C., PARKIN, A. AND SHEPHERD, M., 2004. *Review of impacts of rural land use and management on flood generation. Impact study report. Appendix C Current state of managed rural land and mitigation measures.* R&D Technical Report FD2114/TR. London: Department for Environment, Food and Rural Affairs.

HARROD, T.R. AND THEURER, F.D., 2002. Sediment. In *Agriculture, Hydrology and Water Quality* (ed. P.M. Haygarth and S.C. Jarvis), pp. 155-170. Wallingford: CABI Publishing.

HARROLD, L.L., BRAKENSIEK, D.L., AND MCGUINNESS, J.L., 1962. *Influence of land use and treatment on the hydrology of small watersheds at Coshocton, Ohio 1938-1957.* Technical Bulletin 1256. Washington DC: US Department of Agriculture.

HARTWICH, J., SCHMIDT, M., BOLSCHER, J., RHIENHARDT-IMJELA, C., MURACH D. AND SCHULTE, A., 2016. Hydrological modelling of changes in the water balance due to the impact of woody biomass production in the North German Plain. *Environmental Earth Sciences*, 75, 1071.

HATTERMANN, F.F., KRYSANOVA, V. and HESSE, C., 2008. Modelling wetland processes in regional applications. *Hydrological Sciences Journal-Journal Des Sciences Hydrologiques*, 53 (5), 1001-1012.

HAYGARTH, P.M. AND JARVIS, S.C., 1999. Transfer of phosphorus from agricultural soils. *Advances in Agronomy*, 66, 196-249.

HEATHWAITE, A.L. AND JOHNES, P.J., 1996. The contribution of nitrogen species and phosphorus fractions to stream water quality in agricultural catchments. *Hydrological Processes*, 10 (7), 971–983.

HEATHWAITE, A.L., BURT, T.P. AND TRUDGILL, S.L., 1989. Runoff, sediment, and solute delivery in agricultural drainage basins. In *Regional Characterization of Water Quality* (ed. S. Ragone), pp. 175-190. IAHS Publication No. 182. Walllingford: IAHS Press.

HEATHWAITE, A.L., BURT, T.P. AND TRUDGILL, S.T., 1990. Land-use controls on sediment production in a lowland catchment, south-west England. In *Soil Erosion on Agricultural Land* (ed. J. Boardman. I.D.L. Foster and J.A. Dearing), pp. 70-86. Chichester: John Wiley and Sons.

HEATHWAITE, A.L., P.F. QUINN, P.F. AND HEWETT, C.J.M., 2005. Modelling and managing critical source areas of diffuse pollution from agricultural land using flow connectivity simulation. *Journal of Hydrology*, 304 (1-4): 446-461

HEDGELINK UK, 2013. The complete hedge good management guide. Available from: <u>http://www.hedgelink.org.uk/cms/cms_content/files/30_complete_good_hedge_manag</u> <u>ement_guide_leaflet.pdf</u> [Accessed 18 July 2017].

HEIMANN, D.C. and KREMPA, H.M., 2011. Cumulative effects of impoundments on the hydrology of riparian wetlands along the Marmaton River, West-Central Missouri, USA. *Wetlands*, 31 (1), 135-146.

HEIN, T., SCHWARZ, U., HABERSACK, H., NICHERSU, I., PREINER, S., WILLBY, N. AND WEIGELHOFER, G., 2016. Current status and restoration options for floodplains along the Danube River. *Science of the Total Environment*, 543, 778-790.

HENDERSON, T., 2015. *Haltwhistle flood protection scheme inspired by children's game Kerplunk* [online]. Available from: <u>http://www.chroniclelive.co.uk/news/north-east-news/haltwhistle-flood-protection-scheme-inspired-9942573</u> [Accessed 6 July 2017].

HEWLETT, J.D. AND HIBBERT, A.R., 1963. Moisture and energy conditions within a sloping soil mass during drainage. *Journal of Geophysical Research*, 68 (4), 1081-1087.

HEWLETT, J.D. and HIBBERT, A.R., 1967. Factors affecting the response of small watersheds to precipitation in humid areas. In *Forest Hydrology* (ed. W.E. Sopper and H.W. Lull), pp. 275-290. New York: Pergamon Press.

HEWLETT, J.D. AND HELVEY, J.D., 1970. Effects of forest clear-felling on the storm hydrograph. *Water Resources Research*, 6 (3), 768-782.

HEY, R.D. AND THORNE, C.R., 1986. Stable channels with mobile gravel beds. *Journal of Hydraulic Engineering*, 112 (8), 671-689.

HICKMANN, M., ASH, J. AND FENN, T., 2001. *Sustainable flood defence: the case for washlands*. English Nature Research Reports No. 406. Peterborough: English Nature.

HILL, P.R. AND MANNERING, J.V., 1995. *Conservation tillage and water quality*. WQ-20. West Lafayette, IN: School of Agriculture, Purdue University

HODGE, I. AND READER, M., 2010. The introduction of Entry Level Stewardship in England: extension or dilution in agri-environment policy. *Land Use Policy*, 27 (2), 270-282

HILLMAN, G.R., 1998. Flood wave attenuation by a wetland following a beaver dam failure on a second order boreal stream. *Wetlands*, 18 (1), 21-34.

HOFSTEDE, J.L.A., 2003. Integrated management of artificially created salt marshes in the Wadden Sea of Schleswig-Holstein, Germany. *Wetlands Ecology and Management*, 11 (3), 183-194.

HOLDEN, J., 2006. Sediment and particulate carbon removal by pipe erosion increase over time in blanket peatlands as a consequence of land drainage. *Journal of Geophysical Research*, 111 (F2), 1-6.

HOLDEN, J., CHAPMAN, P.J. AND LABADZ, J.C., 2004. Artificial drainage of peatlands: hydrological and hydrochemical process in wetland restoration. *Progress in Physical Geography*, 28 (1), 95-123.

HOLDEN, J., KIRKBY, M.J., LANE, S.N., MILLEDGE, D.G., BROOKES, C.J., HOLDEN, V. AND MCDONALD, A.T., 2008. Overland flow velocity and roughness properties in peatlands. *Water Resources Research*, 44 (6), article W06415.

HOLDEN, J., WALLAGE, Z. E., LANE, S. N. AND MCDONALD, A. T., 2011. Water table dynamics in undisturbed, drained and restored blanket peat. *Journal of Hydrology*, 402(1), 103-114.

HOLDEN, J., GREEN, S.M., BAIRD, A.J., GRAYSON, R.P., DOOLING, G.P., CHAPMAN, P.J., EVANS, C.D., PEACOCK, M. AND SWINDLES, G., 2016. The impact of ditch blocking on the hydrological functioning of blanket peatlands. *Hydrological Processes*, 31 (3), 525-539.

HOLLAND, J.M., 2004. The environmental consequences of adopting conservation tillage in Europe: reviewing the evidence. *Agriculture, Ecosystems and Environment*, 103 (1), 1-25.

HOLM, G.O., PEREZ, B.C., MCWHORTER, D.E., KRAUSS, K.W., JOHNSON, D.J., RAYNIE, R.C. AND KILLEBREW, C.J., 2016. Ecosystem level methane fluxes from tidal freshwater and brackish marshes of the Mississippi River Delta: implications for coastal wetland carbon projects. *Wetlands*, 36 (3), 401-413.

HOLMAN, I.P., HOLLIS, J.M., BRAMLEY, M.E., AND THOMPSON, T.R.E., 2003. The contribution of soil structural degradation to catchment flooding: a preliminary investigation of the 2000 floods in England and Wales. *Hydrology and Earth System Sciences*, 7 (5), 755-766.

HOLMES, N. AND RAVEN, P., 2014. *Rivers: a natural and not-so-natural history*. Oxford: British Wildlife Publishing.

HORNBECK, J.W., 1973, Storm flow from hardwood-forested and cleared watersheds in New Hampshire. *Water Resources Research*, 9 (2), 346-354.

HOUSE, A.R., THOMPSON, J.R., SORENSEN, J.P.R., ROBERTS, C. and ACREMAN, M.C., 2016. Modelling groundwater/surface water interaction in a managed riparian chalk valley wetland. *Hydrological Processes*, 30 (3), 447-462.

HOUSTON, J., ROONEY, P.AND DOODY, P., 2008. *The conservation and management of coastal vegetated shingle in England*. Report of the meeting at Salthouse, north Norfolk on 18 September 2008. Sand Dune and Shingle Network Occasional Paper No. 1. Liverpool: Liverpool Hope University; Sheffield: Natural England.

HUDSON, N., 1995. Soil Conservation, 3rd edition. London: B.T. Batsford.

HUDSON, P.F., MIDDELKOOP, H. and STOUTHAMER, E. 2008. Flood management along the Lower Mississippi and Rhine Rivers (the Netherlands) and the continuum of geomorphic adjustment. *Geomorphology*, 101 (1-2), 209-236.

HUDSON, P.F., HEITMULLER, F.T. AND LEITCH, M.B., 2012. Hydrologic connectivity of oxbow lakes along the lower Guadalupe River, Texas: the influence of geomorphic

and climatic controls on the 'flood pulse concept'. *Journal of Hydrology*, 414-415, 174-183.

HUGHES, F.M.R. (ED.), 2003. *The Flooded Forest: Guidance for policy makers and river managers in Europe on the restoration of floodplain forests.* Cambridge: FLOBAR2, Department of Geography, University of Cambridge.

HUGHES, F.M.R., ADAMS, W.M., MULLER, E., NILSSON, C., RICHARDS, K.S.R., BARSOUM, N., DECAMPS, H., FOUSSADIER, R., GIREL, J., GUILLOY, H., HAYES, A., JOHANSSON, M., LAMBS, L., PAUTOU, G., PEIRY, J.-L., PERROW, M., VAUTIER, F. AND WINFIELD, M., 2001. The importance of different scale processes for the restoration of floodplain woodlands. *Regulated Rivers: Research and Management*, 17 (4-5), 325-345.

HYGELUND, B. AND MANGA, M., 2003. Field measurements of drag coefficients for model large woody debris. *Geomorphology*, 51 (1-3), 175-185.

IROUMÉ, A., AND HUBER, A., 2002. Comparison of interception losses in a broadleaved native forest and a *Pseudotsuga menziesii* plantation in the Andes Mountains of southern Chile. *Hydrological Processes*, 16 (12), 2347-2361.

IROUMÉ, A., MAYEN, O. AND HUBER, A., 2006. Runoff and peak flow responses to timber harvest and forest age in southern Chile. *Hydrological Processes*, 20 (1), 37-50.

IROUMÉ, A., PALACIOS, H., BATHURST, J.C., AND HUBER, A., 2010. Runoff and peakflows after clearcutting and the establishment of a new plantation in an experimental catchment, southern Chile. *Bosque*, 31, 117-128.

JACOBS, 2011. *Development of methodology for assessment required under Section* 20. Stirling: Scottish Environment Protection Agency.

JÄHNIG, S. C., BRABEC, K., BUFFAGNI, A., ERBA, S., LORENZ, A. W., OFENBÖCK, T., VERDONSCHOT, P. F. M. and HERING, D., 2010. A comparative analysis of restoration measures and their effects on hydromorphology and benthic invertebrates in 26 central and southern European rivers. *Journal of Applied Ecology*, 47, 671-680.

JACKSON, B. M., WHEATER, H. S., MCINTYRE, N. R., CHELL, J., FRANCIS, O. J., FROGBROOK, Z. AND SOLLOWAY., 2008. The impact of upland land management on flooding: insights from a multiscale experimental and modelling programme. *Journal of Flood Risk Management*, 1(2), 71-80.

JBA CONSULTING, 2007. *Ripon Land Management Project*. Final report to Defra (Project SLD2332). Skipton: JBA Consulting.

JBA CONSULTING, 2015. *Woodland and Natural Flood Management – lessons learnt*. Farnham, Surrey: Forestry Commission.

JEFFRIES, R., DARBY, S.E. AND SEAR, D.A., 2003. The influence of vegetation and organic debris on flood-plain sediment dynamics: case study of a low-order stream in the New Forest, England. *Geomorphology*, 51 (1-3), 61-80.

JOHNSON, R.C. (ED.), 1995. *Effects of upland afforestation on water resources. The Balquhidder Experiment 1981-1991.* Report No.116. Wallingford: Institute of Hydrology.

JNCC, 2011. *Towards an assessment of the state of UK peatlands*. JNCC Report No. 445. Peterborough: Joint Nature Conservation Committee.

JNCC, 2016. *Extent and distribution of UK coastal habitats* [online]. Peterborough: Joint Nature Conservation Committee. Available from: <u>http://jncc.defra.gov.uk/page-5379-theme=default</u> [Accessed 21 July 2017].

JONCZYK, J., QUINN, P.F., RIMMER, D.L., BURKE, S. and WILKINSON, M., 2008. *Farm Integrated Runoff Management (FIRM) plans: a tool to reduce diffuse pollution.* Presentation to British Hydrological Society 10th National Hydrology Symposium (Exeter, 2008).

JONES, J.A., 2000. Hydrologic processes and peak discharge response to forest removal, regrowth, and roads in 10 small experimental basins, western Cascades, Oregon. *Water Resources Research*, 36 (9), 2621-2642.

JONES, M. 2010. *Farming Floodplains for the Future: final report 2007-2010*. Stafford: Staffordshire Wildlife Trust.

JUNK, W.J., BAYLEY, P.B. AND SPARKS, R.E., 1989. The flood pulse concept in river–floodplain systems. *Canadian Special Publication of Fisheries and Aquatic Sciences*, 106, 110-127.

KADYKALO, A.N. AND FINDLAY, S.C., 2016. The flow regulation services of wetlands. *Ecosystems Services*, 20, 91-103.

KAMPHORST, E.C., JETEN, V., GUÉRIF, J., PITKÄNEN, J., IVERSEN, B.V., DOUGLAS, J.T. AND PAZ, A. 2000. Predicting depressional storage from soil surface roughness. *Soil Science Society of America Journal*, 64 (5), 1749-1758.

KAIL, J., HERING, D., MUHAR, S., GERHARD, M. and PREIS, S. 2007. The use of large wood in stream restoration: experiences from 50 projects in Germany and Austria. *Journal of Applied Ecology*, 44 (6), 1145-1155.

KAIL, J., BRABEC, K., POPPE, M. and JANUSCHKE, K., 2015. The effect of river restoration on fish, macroinvertebrates and aquatic macrophytes: A meta-analysis. *Ecological Indicators*, 58, 311-321.

KAY, C., WYER, M., STAPLETON, C., EDWARDS, A., DAVIES, C., FRANCIS, C., WATKINS, J., MATTHEWS, R., CLARKE, A., DAVIES, O., MCDONALD, A., KAY, D., 2010. Environmental benefits and impacts of farm ponds and wetlands: some preliminary findings from Scotland and Wales. In *Climate, Water and Soil: Science, Policy and Practice*, Proceedings of the SAC and SEPA Biennial Conference (Edinburgh, 2010) edited by K. Crighton and R. Audsley, pp 111-145. Edinburgh: Scottish Agricultural College.

KEESSTRA, S., KONDRLOVA, E., CZAJKA, A., SEEGER, M. and MAROULIS, J. 2012. Assessing riparian zone impacts on water and sediment movement: a new approach. *Netherlands Journal of Geosciences*, 91 (1-2), 245-255.

KELLETT, A.J., 1978. *Poaching of grassland and the role of drainage*. Technical Report 78/1. London: Ministry of Agriculture, Fisheries and Food, Field Drainage Experimental Unit.

KEMP, D.R. AND MICHALK, D.L., 2007. Towards sustainable grassland and livestock management. *The Journal of Agricultural Science*, 145 (6), 543-564.

KERETSZ, A., MADARASZ, B., CSEPINSKY, B. AND BENKE, S., 2010. The role of conservation agriculture in landscape protection, *Hungarian Geographical Bulletin*, 59 (2), 167-180.

KLEIJN, D., BAQUERO, R.A., CLOUGH, Y., DIAZ, M., ESTEBAN, J.D., FERNÁNDEZ, F., GABRIEL, D., HERZOG, F., HOLZSCHUH, A., JÖHL, R. AND KNOP, E., 2006. Mixed biodiversity benefits of agri-environment schemes in five European countries. *Ecology Letters*, 9 (3), 243-254.

KITE, G., 2001. Modelling the Mekong: hydrological simulation for environmental impact studies. *Journal of Hydrology*, 253 (1-4), 1-13.

KITTS, D.R., 2010. *The hydraulic and hydrological performance of large wood accumulations in a low-order forest stream.* PhD thesis, University of Southampton.

KOBELL, R. AND JOURNAL, B., 2015. Farmers reduce pollution after ditching old way of handling runoff [online]. South Bend, IN: Environmental Change Initiative. Available from: <u>http://environmentalchange.nd.edu/news-events/news/farmers-reduce-pollution-after-ditching-old-way-of-handling-runoff/</u> [Accessed 4 July 2017].

KONDOLF, G.M., 1998. Lessons learned from river restoration projects in California. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 8 (1), 39-52.

KONDOLF, G.M., 2006. River restoration and meanders. *Ecology and Society*, 11 (2), article 42.

KONDOLF, G.M., BOULTON, A.J., O'DANIEL, S., POOLE, G.C., RAHEL, F.J., STANLEY, E.H., WOHL, E., BÅNG, A., CARLSTROM, J., CRISTONI, C., HUBER, H., KOLJONEN, S., LOUHI, P. AND NAKAMURA, K., 2006. Process-based ecological river restoration: visualizing three-dimensional connectivity and dynamic vectors to recover lost linkages. *Ecology and Society*, 11 (2), article 5.

KOONTZ, M.B., PEZESHKI, S.R. AND DELAUNE, R.D., 2014. Nutrient dynamics in a restored wetland. *Communications in Soil Science and Plant Analysis*, 45 (5), 609-623.

KOUWEN, N. 2013. WATFLOOD®/WATROUTE® hydrological model routing and flow forecasting system. Waterloo, Ontario: University of Waterloo.

KRAUSE, S., KLAAR, M.J., HANNAH, D.M., MANT, J., BRIDGEMAN, J., TRIMMER, M. AND MANNING-JONES, S., 2014. The potential of large woody debris to alter biogeochemical processes and ecosystem services in lowland rivers. *Wiley Interdisciplinary Reviews: Water*, 1 (3), 263-275.

KRAVČIK, M., KOHUTIAR, J., GAŽOVIČ, M., KOVÁČ, M., HRIB, M., ŠUTY, P. AND KRAVČIKOVÁ, D., 2012. *After us, the desert and the deluge?* Košice, Slovakia: NGO People and Water.

KREIS, N., LEVIANDIER, T. AND ARNAUD, P., 2005. Hydrological and hydraulics modelling to assess mitigation effectiveness of floodplain reconnection in the context of a mountain river. *International Journal of River Basin Management*, 3 (2), 117-123.

KRONVANG, B., SVENDSEN, L., BROOKES, A., FISHER, K., MÖLLER, B., OTTOSEN, O., NEWSON, M. AND SEAR, D., 1998. Restoration of the rivers Brede, Cole and Skerne: a joint Danish and British EU-LIFE demonstration project. III – Channel morphology, hydrodynamics and transport of sediment and nutrients. *Aquatic Conservation. Marine and Freshwater Ecosystems*, 8 (1), 209-222.

KRONVANG, B., WIBORG, I., HECKRATH, G. AND BAATTRUP-PEDERSEN, A., 2010. Multifunktionelle randzoner for natur, miljø og friluftsliv [Multifunctional buffer strips for nature, environment and leisure]. *Jord og Viden*, 155 (9), 12-15.

KRONVANG, B., ANDERSEN, H.E., JENSEN, P.N., HECKRATH, G.J., RUBÆK, G.H. AND KJÆRGAARD, C., 2011. *Effekt på fosforudledning af 10 m brede randzoner* [Effect of 10m buffer strips on phosphorus losses]. Memorandum No. 34166. Aarhus, Denmark: Danish Environmental Protection Ministry.

KUNDZEWICZ, Z.W. and MENZEL, L., 2005. Natural flood reduction strategies – a challenge. *International Journal of River Basin Management,* 3 (2), 125-131.

KURAŚ, P.K., ALILA, Y., AND WEILER, M., 2012. Forest harvesting effects on the magnitude and frequency of peak flows can increase with return period. *Water Resources Research*, 48 (1), article W01544.

KUTIJA, V. AND MURRAY M.G., 2007. An object-oriented approach to the modelling of free-surface flows. *Journal of Hydroinfomatics*, 9 (2) 81-94.

LAKE, P.S., 2012. Flows, floods, floodplains and river restoration. *Ecological Management and Restoration*, 13 (3), 210-211.

LAL, R., REICOSKY, D. C. AND HANSON, J. D., 2007. Evolution of the plow over 10,000 years and the rationale for no-till farming. *Soil and Tillage Research*, 93(1), 1-12.

LAMANDÉ, M., HALLAIRE, V., CURMIA, P., PERES, G. AND CLUZEAU, D., 2003. Changes of pore morphology, infiltration and earthworm community in a loamy soil under different agricultural managements. *Catena*, 54 (3), 637-649. LAMOUROUX, N., GORE, J.A., LEPORI, F. and STATZNER, B., 2015. The ecological restoration of large rivers needs science-based, predictive tools meeting public expectations: an overview of the Rhône project. *Freshwater Biology*, 60 (6), 1069-1084.

LANE, S.N., 2001. More floods, less rain? Changing hydrology in a Yorkshire context. *The Regional Review*, 12, 18-19.

LANE, S.N., 2008. Slowing the floods in the UK Pennine Uplands: A case of waiting for Godot. *Journal of Practical Ecology and Conservation*, 7 (1), 75-91.

LANE, S.N., 2017. Natural flood management. *Wiley Interdisciplinary Reviews: Water*, 4 (3), e1211.

LANE, S.N AND MILLEDGE, D.G., 2013. Impacts of upland open drains upon runoff generation: a numerical assessment of catchment-scale impacts. *Hydrological Processes*, 27 (12), 1701-1726.

LANE, S.N., MORRIS, J., O'CONNELL, P.E. AND QUINN, P.F., 2007. Managing the rural landscape. In *Future Flooding and Coastal Erosion Risks* (ed. C.R. Thorne, E.P. Evans and E.C. Penning-Rowsell), Chapter 18, pp. 297-319. London: Thomas Telford.

LANE, S.N., REANEY, S.M. AND HEATHWAITE, A.L., 2009. Representation of landscape hydrological connectivity using a topographically driven surface flow index. *Water Resources Research*, 45 (8), article W08423.

LANGFORD, T.E.L., LANGFORD, J. and HAWKINS, S.J., 2012. Conflicting effects of woody debris on stream fish populations: implications for management. *Freshwater Biology*, 57 (5), 1096-1111.

LAROSE, S., PRICE, J. AND ROCHEFORT, L., 1997. Rewetting of a cutover peatland: hydrologic assessment. *Wetlands*, 17, 416-423.

LATHAM, D., PONTEE, N., MONROE, B. AND BOND, I., 2013. *Reconstructing habitats in a heavily industrialised estuary: from brine wells to salt marsh.* Presentation at Coasts, Marine Structures and Breakwaters 2013, Aberdeen, 17-20 September 2013

LEITE RIBEIRO, M., BLANCKAERT, K. and SCHLEISS, A., 2015. Local tributary widening for river rehabilitation. *Ecohydrology. DOI*, 10.

LEOPOLD, L.B. AND MADDOCK, T., 1954. *The Flood Control Controversy*. New York: Ronald Press.

LEVASSEUR, F.. BAILLY, J.S., LAGACHERIE, P., COLIN, F. AND RABOTIN M., 2012. Simulating the effects of spatial configurations of agricultural ditch drainage networks on surface runoff from agricultural catchments. *Hydrological Processes*, 26 (22), 3393-3404.

LEWIN, J., 2013. Enlightenment and the GM floodplain. *Earth Surface Processes and Landforms*, 38 (1), 17-29.

LEWIN, J. AND ASHWORTH, P.J., 2014. The negative relief of large river floodplains. *Earth-Science Reviews*, 129, 1-23.

LIN, Y. AND WEI, X., 2008. The impact of large-scale forest harvesting on hydrology in the Willow watershed of Central British Columbia. *J. Hydrol.* 359 (1–2), 141–149.

LINDSAY, J.B., CREED, I.F. AND BEALL, F.D., 2004. Drainage basin morphometrics for depressional landscapes. *Water Resources Research*, 40 (9), article W09307.

LINDSAY, R., BIRNIE, R. AND CLOUGH, J., 2014a. *Peat bog ecosystems: key definitions.* IUCN UK Committee Peatland Programme Briefing Note No. 1. Edinburgh: International Union for the Conservation of Nature UK Peatland Programme.

LINDSAY, R., BIRNIE, R. AND CLOUGH, J., 2014b. *Impacts of artificial drainage on peatlands*. IUCN UK Committee Peatland Programme Briefing Note No. 3. Edinburgh: International Union for the Conservation of Nature UK Peatland Programme.

LINDSAY, R., BIRNIE, R. AND CLOUGH, J., 2014c. Peat *bog ecosystems: structure, form, state and condition.* IUCN UK Committee Peatland Programme Briefing Note No. 2. Edinburgh: International Union for the Conservation of Nature UK Peatland Programme.

LINDSAY, R., BIRNIE, R AND CLOUGH, J., 2016. *Peatland restoration*. IUCN UK Committee Peatland Programme Briefing Note No. 11. Edinburgh: International Union for the Conservation of Nature UK Peatland Programme.

LIU, X., ZHANG, X. AND ZHANG, M., 2008. Major factors influencing the efficacy of vegetated buffers on sediment trapping: A review and analysis. *Journal of Environmental Quality*, 37(5), 1667-1674.

LIU, G., ZHANG, J., TIAN, G. AND WEI, C., 2005. The effects of land use on purplish soil erosion in hilly area of Sichuan Province China. *Journal of Mountain Science – English* 2, (1), 68-75.

LIU, Y., GEBREMESKEL, S., DE SMEDT, F., HOFFMANN, L. AND PFISTER, L., 2004. Simulation of flood reduction by natural river rehabilitation using a distributed hydrological model. *Hydrology and Earth System Sciences Discussions*, 8 (6), 1129-1140.

LOFROTH, E., 1998. The dead wood cycle. In *Conservation Biology Principles for Forested Landscapes* (ed. J. Voller and S. Harrison), pp. 185-207. Vancouver: UBC Press.

LONDON BOROUGH OF BARKING AND DAGENHAM, 2012. *Welcome to Mayesbrook Park. Britain's first climate change park.* Publication Ref. No. MC6826. London: London Borough of Barking and Dagenham. Available from: https://www.lbbd.gov.uk/wp-content/uploads/2014/07/MayesbrookParkLeaflet.pdf [Accessed 6 July 2017].

L'UBOMÍR, S., FERANEC, J. AND NOVÁČEK, J., 2011. Land cover changes in small catchments in Slovakia during 1990–2006 and their effects on frequency of flood events. *Natural Hazards*, 56 (1), 195-214.

LÜDERITZ, V., SPEIERL, T., LANGHEINRICH, U., VÖLKL, W. and GERSBERG, R. M., 2011. Restoration of the Upper Main and Rodach rivers – The success and its measurement. *Ecological Engineering*, 37, 2044-2055.

LYNCH, K., JACKSON, D.W.T. AND COOPER, J.A.G., 2009. Foredune accretion under offshore winds. *Geomorphology*, 105 (1-2), 139-146.

MACDONALD, A.J., POULTON, P.R., HOWE, M.T., GOULDING, K.W.T. AND POWLSON, D.S., 2005. The use of cover crops in cereal-based cropping systems to control nitrate leaching in SE England. *Plant and Soil*, 273 (1), 355-373.

MACDONALD, A., LAPWORTH, D., HUGHES, A., AUTON, C., MAURICE, L., FINLAYSON, A. AND GOODDY, D., 2014. Groundwater, flooding and hydrological functioning in the Findhorn floodplain, Scotland. *Hydrology Research*, 45 (6), 755-773.

MACLEOD, C.J.A., HUMPRHREYS, M.W., WHALLEY, W.R., TURNER, L., BINLEY, A., WATTS, C.W., SKØT, L., JOYNES, A., HAWKINS, S., KING, I.P., O'DONOVAN, S AND HAYGARTH, P., 2013. A novel grass hybrid to reduce flood generation in temperate regions. *Scientific Reports*, 3, article 1683.

MAFF, 1970. Modern farming and the soil. Report of the Agricultural Advisory Council on soil structure and soil fertility (the Strutt Report). London: HMSO.

MAINSTONE, C.P. AND HOLMES, N.T., 2010. Embedding a strategic approach to river restoration in operational management practice experiences in England. *Aquatic Conservation*, 20, 82e95.

MALTBY, E., ACREMAN, M., BLACKWELL, M., EVERARD, M. AND MORRIS, J., 2013. The challenges and implications of linking wetland science to policy in agricultural landscapes – experience from the UK National Ecosystem Assessment. *Ecological Engineering*, 56, 121-133.

MANNERS, R.B. AND DOYLE, M.W., 2008. A mechanistic model of woody debris jam evolution and its application to wood-based restoration and management. *River Research and Applications*, 24 (8), 1104-1123.

MANNING, R., 1891. On the flow of water in open channels and pipes. *Transactions of the Institution of Civil Engineers of Ireland*, 20, 161-207.

MAPA, R.B., 1995. Effect of reforestation using *Tectona grandis* on infiltration and soil water retention. *Forest Ecology and Management*, 77 (1-3), 199-125.

MARC, V. AND ROBINSON, M., 2007. The long-term water balance (1972-2004) of upland forestry and grassland at Plynlimon, mid-Wales. *Hydrology and Earth System Sciences*, 11 (1), 44-60.

MARSHALL, M.R., BALLARD, C.E., FROGBROOK, Z.L., SOLLOWAY, I., MCINTYRE, N., REYNOLDS, B. AND WHEATER, H.S., 2014. The impact of rural land management changes on soil hydraulic properties and runoff processes: results from experimental plots in upland UK. *Hydrological Processes*, 28 (4), 2617-2629.

MARSTON, R.A., 1982. The geomorphic significance of log steps in forest streams. *Annals of the Association of American Geographers*, 72 (1), 99-108.

MARTINEZ, M.L., GALLEGO-FERNÁNDEZ, J.B. AND HESP, P.A. (ED.), 2013. *Restoration of Coastal Dunes*. Dordrecht: Springer.

MASSELINK, G., HUGHES, M. AND KNIGHT, J. (ED.), 2014. *Introduction to Coastal Processes and Geomorphology*, 2nd edition. London: Hodder Education.

MAYOR, J. AND HICKS, C., 2009. Potential impacts of elevated CO₂ on plant interactions, sustained growth, and carbon cycling in salt marsh ecosystems. In *Human Impacts on Salt Marshes: A Global Perspective* (ed. B.R. Silliman, T. Grosholz and M.D. Bertness), pp. 207-228. Berkeley, CA: University of California Press

MCCLAIN, M.E., BOYER, E.W., DENT, C.L., GERGEL, S.E., GRIMM, N.B., GROFFMAN, P.M., HART, S.C., HARVEY, J.W., JOHNSTON, C A. and MAYOR, G.A.E., 2003. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. *Ecosystems*, 6 (4), 301-312.

MCCULLOCH, J.S.G. AND ROBINSON, M., 1993. History of forest hydrology. *Journal of Hydrology*, 150 (2-4), 189-216.

MCGLOTHIN, D., JACKSON, W.J. and SUMMERS, P., 1988. Groundwater geomorphic processes and riparian values: San Pedro River, Arizona. Proceedings of a Symposium on Water-Use data for Water resource Management, 28-31 August 1988, Tuscan AZ. Pp 537-545. American Water Resources Association, Tuscon, AZ.

MCGONIGLE, D.F., BURKE, S.P., COLLINS, A.L., GARTNER, R., HAFT, M.R., HARRIS, R.C. AND LOVETT, A.A., 2014. Developing Demonstration Test Catchments as a platform for transdisciplinary land management research in England and Wales. *Environmental Science: Processes and Impacts*, 16 (7), 1618-1628.

MCINTYRE, N. AND MARSHALL, M., 2010. Identification of rural land management signals in runoff response. *Hydrological Processes*, 24 (24), 3521-3534.

MCINTYRE, N. AND THORNE, C. (ed.), 2013. Land use management effects on flood flows and sediments – guidance on prediction. CIRIA Report C719. London: CIRIA.

MCINTYRE, N., BALLARD, C., BULYGINA, N., FROGBROOK, Z., CLUCKIE, I., DANGERFIELD, S., EWEN, J., GERIS, J., HENSHAW, A., JACKSON, B., MARSHALL, M., PAGELLA, T., PARK, J.-S., REYNOLDS, B., O'CONNELL, E., O'DONNELL, G., SINCLAIR, F., SOLLOWAY, I., THORNE, C. AND WHEATER, H., 2012. The potential for reducing flood risk through changes to rural land management: outcomes from the Flood Risk Management Research Consortium. Presentation to British Hydrological Society's 11th National Symposium, *Hydrology for a Changing World* (Dundee, 2012).

MCPFE, 2007. *Warsaw Resolution 2: Forests and water*. In Proceedings of Fifth Ministerial Conference for the Protection of Forestry in Europe (5th MCPRE) (Warsaw, 2007), pp. 207-209. Warsaw: Ministerial Conference for the Protection of Forestry in Europe.

MELANDER, B., NØRREMARK, M. AND KRISTENSEN, E. F., 2013. Combining mechanical rhizome removal and cover crops for Elytrigia repens control in organic barley systems. *Weed Research*, 53(6), 461-469.

MELLOR, C., 2014. Monetising the value of ecosystem services provided by river restoration projects. 2014 RRC conference.

METCALFE, P., BEVEN, K. AND FREER, J., 2015. Dynamic TOPMODEL: a new implementation in R and its sensitivity to time and space intervals. *Environmental Modelling and Software*, 72 (C), 155-172.

METCALFE, P., BEVEN, K., HANKIN, B. and LAMB, R., 2016. *Natural Flood Management in context: evaluating and enhancing the impact*. EGU General Assembly Conference Abstracts 2016, abstract 8831.

METCALFE, P., BEVEN, K., HANKIN, B. AND LAMB, R., 2017. A modelling framework for evaluation of the hydrological impacts of nature-based approaches to flood risk management, with application to in-channel interventions across a 29 km 2 scale catchment in the United Kingdom: A modelling framework for nature-based flood risk management. *Hydrological Processes*, 31 (9), 1734-1748.

MET OFFICE., 2016. Soil water deficit maps for Great Britain. Met Office, Exeter, Devon.

MITSCH, W.J. AND DAY, J.W., 2006. Restoration of wetlands in the Mississippi–Ohio– Missouri (MOM) River Basin: Experience and needed research. *Ecological Engineering*, 26 (1), 55-69.

MITSCH, W.J. AND GOSSELINK, J.G., 1993. *Wetlands*, 2nd edition. New York: Van Nostrand Reinhold.

MITSCH, W.J. AND GOSSELINK, J.G., 2007. *Wetlands*, 4th edition. New York: John Wiley and Sons.

MÖLLER, I. AND SPENCER, T., 2002. Wave dissipation over macro-tidal saltmarshes: effects of marsh edge typology and vegetation change. *Journal of Coastal Research*, Special Issue 36, 506-521.

MÖLLER, I., SPENCER, T. AND FRENCH, J.R., 1996. Wind wave attenuation over saltmarsh surfaces: preliminary results from Norfolk, England. *Journal of Coastal Research*, 12 (4), 1009-1016.

MÖLLER, I., SPENCER, T., FRENCH, J.R., LEGGETT, D.J. AND DIXON, M, 2001. The sea-defence value of salt marshes: field evidence from north Norfolk. *Water and Environment Journal*, 15 (2), 109-116.

MÖLLER, I., KUDELLA, M., RUPPRECHT, F., SPENCER, T., PAUL, M., VAN WESENBEECK, B.K., WOLTERS, G., JENSEN, K., BOUMA, T.J., MIRANDA-LANGE, M. AND SCHIMMELS, S., 2014. Wave attenuation over coastal salt marshes under storm surge conditions. *Nature Geoscience*, 7, 727-731.

MONTGOMERY, D.R., BUFFINGTON, J.M., SMITH, R.D., SCHMIDT, K.M. AND PESS, G. 1995. Pool spacing in forest channels. *Water Resources Research*, 31 (4), 1097-1105.

MONTGOMERY, D.R., COLLINS, B.D., BUFFINGTON, J.M. and ABBE, T.B., 2003. Geomorphic effects of wood in rivers. In *Ecology and Management of Wood in World Rivers*, Proceedings of the International Conference on Wood in World Rivers (Corvallis, OR, 2000), American Fisheries Society Symposium 37 (ed. S.V. Gregory, K.L. Boyer and A.M. Gurnell), pp. 21-47. Bethseda, MD: American Fisheries Society.

MOORS FOR THE FUTURE, 2008. *Geo-textiles*. Factsheet. Castleton, Derbyshire: Moors for the Future. http://www.moorsforthefuture.org.uk/geo-textiles [Accessed 7 July 2017].

MORRIS, C. AND POTTER, C., 1995. Recruiting the new conservationists: farmers' adoption of agri-environmental schemes in the UK. *Journal of Rural Studies*, 11 (1), 51-63.

MORRIS, J., BAILEY, A., LEEDS-HARRISON, P., LAWSON, C., ALSOP, D. AND VIVASH, R., 2004a. *Economic dimensions of washland creation in England: a case from Somerset.* Presentation to Ecoflood International Conference, Towards Natural Flood Reduction Strategies (Warsaw, 2003). EcoFlood Network. Available from: <u>http://levis.sggw.waw.pl/ecoflood/contents/articles/s6/html/MorrisEtAl_bz_S6.html</u> [Accessed 7 July 2017].

MORRIS, J., BAILEY, A.P., ALSOP, D., VIVASH, R. AND LAWSON, C., 2004b. Integrating flood management and agri-environment through washland creation in the UK. *Journal of Farm Management*, 12 (1), 33-48.

MORRIS, J., HESS, T.M., GOWING, D., LEEDS-HARRISON, P., BANNISTER, N., WADE, M. AND VIVASH, R., 2004c. *Integrated washland management for flood*

defence and biodiversity. English Nature Research Reports No. 598. Peterborough: English Nature.

MORRIS, J., BAILEY, A., LAWSON, C., LEEDS-HARRISON, P., ALSOP, D. AND VIVASH, R., 2008. The economic dimensions of integrating flood management and agri-environment through washland creation: a case from Somerset, England. *Journal of Environmental Management*, 88 (2), 372-381.

MOTT, N., 2006. *Managing woody debris in river, streams and floodplains.* Stafford: Staffordshire Wildlife Trust.

MOTT, N., 2010. *Fish live in trees too! River rehabilitation and large woody debris.* Stafford: Staffordshire Wildlife Trust.

MOXEY, A. AND MORAN, D., 2014. UK peatland restoration: some economic arithmetic. *Science of the Total Environment*, 484, 114-120

MUSY, A., 2001. Chapter 8. Hydrograph/hydrological processes. In VICAIRE (VIrtual CAmpus In hydrology and water REsources management) project [online]. Lausanne, Switzerland: Ecole Polytechnique Fédérale. Available from: http://echo2.epfl.ch/VICAIRE/mod_1a/chapt_8/main.htm [Accessed 7 July 2017].

MYERS, W.R.C., LYNESS, J.F. AND CASSELLS, J., 2001. Influence of boundary roughness on velocity and discharge in compound river channels. *Journal of Hydraulic Research*, 39 (3), 311-319.

NANSON, G.C. AND CROKE, J.C., 1992. A genetic classification of floodplains. *Geomorphology*, 4 (6), 459-486.

NARAYAN, S., BECK, M.W., REGUERO, B.G., LOSADA, I.J., VAN WESENBEECK, B., PONTEE, N., SANCHIRICO, J.N., INGRAM, J.C., LANGE, G.-M. AND BURKS-COPES, K.A., 2016. The effectiveness, costs and coastal protection benefits of natural and nature-based defences. *PLOS One*, https://doi.org/10.1371/journal.pone.0154735.

NARDINI, A. AND PAVAN, S., 2012. River restoration: not only for the sake of nature but also for saving money while addressing flood risk: a decision-making framework applied to the Chiese River (Po basin, Italy). *Journal of Flood Risk Management*, 5 (2), 111-133.

NASCIMENTO, N.O., ELLIS, J.B., BAPTISTA, M.B. AND DEUTSCH, J.C., 1999. Using detention basins: Operational experience and lessons. *Urban Water*, 1 (2), 113-124.

NATIONAL TRUST 2015. *From source to sea. Natural Flood Management: the Holnicote experience*. Defra Multi-Objective Flood Management Demonstration Project. London: National Trust.

NATIONAL TRUST, undated. *Adapting to change at South Milton Sands* [online]. <u>https://www.nationaltrust.org.uk/south-milton-sands/features/south-milton-sands---adapting-to-change</u> [Accessed August 2016].

NATURAL ENGLAND, 2010a. *Entry Level Stewardship: Environmental Stewardship Handbook*, 3rd edition. Sheffield: Natural England.

NATURAL ENGLAND, 2010b. *Illustrated guide to ponds and scrapes*. Natural England Technical Information Note TIN079. Sheffield: Natural England.

NATURAL ENGLAND, 2013. Mayesbrook Park - Green Infrastructure Case Study: Creating the UK's first climate change park in east London (NE394).

NATURAL ENGLAND, 2015. Managing for ecosystem services. Coastal and marine – managed realignment. In *Ecosystem Services Transfer Toolkit*, Edition 1 [online]. Available from:

http://publications.naturalengland.org.uk/publication/5890643062685696 [Accessed 7 July 2017].

NATURAL ENGLAND, 2016a. *Countryside hedgerows: protection and management* [online]. Available from: <u>https://www.gov.uk/guidance/countryside-hedgerows-regulation-and-management</u> [Accessed 18 July 2017].

NATURAL ENGLAND, 2016b. *Healthy Estuaries 2020: towards addressing coastal squeeze in estuaries.* Improvement Programme for England's Natura 2000 Sites (IPENS): Planning for the Future IPENS002. York: Natural England.

NEARY, D.G., ICE, G.G., AND JACKSON, C.R., 2009. Linkages between forest soils and water quality and quantity. *Forest Ecology and Management*, 258 (10), 2269-2281.

NEPF, H., 1999. Drag, turbulence and diffusion in flow through emergent vegetation. *Water Resources Research*, 35 (2), 479-489.

NERC, 1975. *Flood Studies Report*, 5 volumes. London: Natural Environment Research Council.

NEWCASTLE UNIVERSITY AND ENVIRONMENT AGENCY, 2011. Runoff attenuation features: a guide for all those working in catchment management. Newcastle upon Tyne: Northumbria Regional Flood Defence Committee. Available from: https://research.ncl.ac.uk/proactive/belford/papers/Runoff_Attenuation_Features_Hand book_final.pdf [Accessed 7 July 2017].

NFU, 2017. The flooding manifesto. London: National Farmers' Union.

NGUYEN, M. L., SHEATH, G. W., SMITH, C. M. AND COOPER, A. B., 1998. Impact of cattle treading on hill land: 2. Soil physical properties and contaminant runoff. New Zealand Journal of Agricultural Research, 41(2), 279-290.

NICHOLSON, A.R., 2014. *Quantifying and simulating the impact of flood mitigation features in a small rural catchment*, PhD thesis, Newcastle University. Available from: https://theses.ncl.ac.uk/dspace/bitstream/10443/2382/1/Nicholson,%20A.%2014.pdf [Accessed 7 July 2017].

NICHOLSON, A.R., WILKINSON, M.E., O'DONNELL, G.M. AND QUINN, P.F., 2012. Runoff attenuation features: a sustainable flood mitigation strategy in the Belford catchment, UK. *Area*, 44 (4), 463-469.

NICHOLSON, A., QUINN, P. AND ROSE, S., 2016. Continuing Professional Development (CPD) course on Natural Flood Management (NFM).

NICHOLLS, R.J., COOPER, N.J. AND TOWNSEND, I.H., 2007, The management of coastal flooding and erosion. In *Future Flooding and Coastal Erosion Risks* (ed. C.R. Thorne, E.P. Evans and E.C. Penning-Rowsell), Chapter 23, pp. 392-413. London: Thomas Telford.

NIEHOFF, D., FRITSCH, U. AND BRONSTERT, A., 2002. Land-use impacts on storm runoff generation: scenarios of land-use change and simulation of hydrological response in a meso-scale catchment in SW-Germany. *Journal of Hydrology*, 267 (1-2), 80-93.

NIKORA, V., 2010. Hydrodynamics of aquatic ecosystems: an interface between ecology, biomechanics and environmental fluid mechanics. *River Research and Applications*, 26 (4), 367-384.

NILSSON, C., POLVI, L. E., GARDESTRÖM, J., HASSELQUIST, E. M., LIND, L. and SARNEEL, J. M., 2015. Riparian and in-stream restoration of boreal streams and rivers: success or failure? *Ecohydrology*, 8, 753-764.

NISBET, T.R., 2001. The role of forest management in controlling diffuse pollution in UK forestry. *Forest Ecology and Management*, 143 (1-3), 215-226.

NISBET, T., 2005. *Water use by trees.* Forest Research Information Note FCIN065. Farnham, Surrey: Forest Research.

NISBET, T.R. AND THOMAS, H., 2008. *Restoring floodplain woodland for flood alleviation*. Final report for Defra, Project SLD2316. London: Department for Environment, Food and Rural Affairs.

NISBET, T., MARRINGTON, S., THOMAS, H., BROADMEADOW, S. and VALATIN, G. 2011a. *Slowing the Flow at Pickering. Final report for Phase 1 for Defra FCERM Multi-objective Flood Management Demonstration project RMP5455.* London: Department for Environment, Food and Rural Affairs.

NISBET, T.R., SILGRAM, M., SHAH, N., MORROW, K. AND BROADMEADOW, S., 2011b. *Woodland for water: woodland measures for meeting Water Framework Directive objectives.* Forest Research Monograph 4. Farnham: Surrey: Forest Research.

NISBET, T.R., ROE, P., MARRINGTON, S., THOMAS, H., BROADMEADOW, S.B., AND VALATIN, G., 2015. Slowing the Flow at Pickering. Final report for Phase 1 for Defra FCERM Multi-objective Flood Management Demonstration project RMP5455. London: Department for Environment, Food and Rural Affairs.

NORTH DEVON WILDLIFE TRUST, 2014. *The Culm: a landscape that works.* Exeter: North Devon Wildlife Trust.

NYSSEN, J., PONTZEELE, J. and BILLI, P., 2011. Effect of beaver dams on the hydrology of small mountain streams: example from the Chevral in the Ourthe Orientale basin, Ardennes, Belgium. *Journal of Hydrology*, 402 (1-2), 92-102.

NWRM, 2015a. Synthesis document no 1: Introducing Natural Water Retention Measures: what are NWRM? Brussels: European Commission. Available from: http://nwrm.eu/implementing-nwrm/synthesis-documents [Accessed 7 July 2017].

NWRM, 2015b, *Individual NWRM. N14: Re-naturalization of polder areas.* Brussels: European Commission. Available from: <u>http://nwrm.eu/measure/re-naturalisation-polder-areas</u> [Accessed 13 July 2017].

NWRM, 2015b. *Individual NWRM. U11: Retention ponds*. Brussels: European Commission. Available from: <u>http://nwrm.eu/measure/retention-ponds</u> [Accessed 7 July 2017].

NWRM, 2015c. *Individual NWRM. U04: Swales.* Brussels: European Commission. Available from: <u>http://nwrm.eu/measures-swales</u> [Accessed 7 July 2017].

O'CONNELL, J., 2008. Modelling the effects of floodplain woodland in flood mitigation. A short-term case study. *Irish Forestry*, 65 (1-2), 17-36.

O'CONNOR, K. 1956. The influence of wheel and foot treading on soils under grasslands. *Proceedings of the New Zealand Society of Soil Science*, 2, 35-37.

O'CONNELL, P.E., BEVEN, K.J., CARNEY, J.N., CLEMENTS, R.O., EWEN, J., FOWLER, H., HARRIS, G.L., HOLLIS, J., MORRIS, J., O'DONNELL, G.M., PACKMAN, J.C., PARKIN, A., QUINN, P.F., ROSE, S.C., SHEPHERD, M. AND TELLIER, S., 2004. *Review of impacts of rural land use and management on flood generation. Impact study report.* R&D Technical Report FD2114/TR. London: Department of Environment, Food and Rural Affairs. O'CONNELL, P.E., EWEN, J., O'DONNELL, G. AND QUINN, P., 2007. Is there a link between land-use management and flooding? *Hydrology and Earth Systems Sciences*, 11 (1), pp. 96-107.

OCKENDEN, M., DEASY, C., QUINTON, J., BAILEY, A., SURRIDGE, B. AND STOATE, C., 2012. Evaluation of field wetlands for mitigation of diffuse pollution from agriculture: Sediment retention, cost and effectiveness. *Environmental Science and Policy*, 24, 110-119.

ODONI, N.A. AND LANE, S.N. 2010. Assessment of the impact of upstream land management measures on flood flows in Pickering using OVERFLOW. Contract report to Forest Research for the Slowing the Flow at Pickering project. Durham: Durham University.

OGAWA, H. and MALE, J.W., 1986. Simulating the flood mitigation role of wetlands. *Journal of Water Resources Planning and Management*, 112 (1), 114-128.

O'HARE, M. T., MCGAHEY, C., BISSETT, N., CAILES, C., HENVILLE, P. and SCARLETT, P., 2010. Variability in roughness measurements for vegetated rivers near base flow, in England and Scotland. *Journal of Hydrology*, 385, 361-370.

OLIVER, D.M., HEATHWAITE, A.L., HAYGARTH, P.M. AND CLEGG, C.D., 2005. Transfer of *Escherischia coli* to water from drained and undrained grassland after grazing. *Journal of Environmental Quality*, 34 (3), 918-925.

OPPERMAN, J.J., LUSTER, R., MCKENNEY, B.A., ROBERTS, M. and MEADOWS, A.W., 2010. Ecologically functional floodplains: connectivity, flow regime, and scale. *Journal of the American Water Resources Association*, 46 (2), 211-216.

ORESZCZYN, S., AND LANE, A. 2000. The meaning of hedgerows in the English landscape: different stakeholder perspectives and the implications for future hedge management. *Journal of Environmental Management*, 60 (1), 101-118.

OWEN, G.J., 2016. An assessment of the potential for natural flood management and land management practices to mitigate flooding in catchments. PhD thesis, Newcastle University.

OWEN, G.J., PERKS, M.T., BENSKIN, C.M.H., WILKINSON, M.E., JONCZYK, J., AND QUINN, P.F., 2012. Monitoring agricultural diffuse pollution through a dense monitoring network in the River Eden Demonstration Test Catchment, Cumbria, UK. *Area*, 44 (4), 443-453.

OZDEMIROGLU, E. AND HAILS, R. (ED.), 2016. *Demystifying economic evaluation.* Valuing Nature Programme VNP04. London: Natural Environment Research Council.

PALMER, M., 2012. *Agricultural fine sediment: sources, pathways and mitigation.* PhD thesis, Newcastle University.

PALMER, R.C. AND SMITH, R.P., 2013. Soil structural degradation in SW England and its impact on surface-water runoff generation. *Soil Use and Management*, 29 (4), 567-575.

PALMER, M., ALLAN, J.D., MEYER, J. and BERNHARDT, E.S., 2007. River restoration in the twenty-first century: data and experiential knowledge to inform future efforts. *Restoration Ecology*, 15 (3), 472-481.

PALMER, M. A., MENNINGER, H. L. and BERNHARDT, E., 2010. River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? *Freshwater Biology*, 55, 205-222.

PAOLETTI, M.G., SOMMAGGIO, D., FAVRETTO, M.R., PETRUZZELLI, G., PEZZAROSSA, B. AND BARBAFIERI, M., 1998. Earthworms as useful indicators of agroecosystem sustainability in orchards and vineyards with different inputs. *Applied Soil Ecology*, 10 (1-2), 137-150.

PARK, R., 2013, *Challenges in Wales*. Presentation to ABPmer Habitat Creation Conference (London, 2013).

PARK, J.-S. AND CLUCKIE, I.D., 2006. *Whole catchment modelling project – technical report. The Parrett Catchment Project Report.* PCP/PCP/WEMRC – Technical Report. Nottingham: University of Nottingham, Centre for Environmental Management.

PARK, J.S., REN, Q., CHEN, Y., CLUCKIE, I.D., BUTTS, M. AND GRAHAM, D., 2009. Effectiveness of complex physics and DTM based distributed models for flood risk management of the River Tone, UK. *IAHS Publ. no. 33*1, 114–121. (ISBN: 978-1-907161-02-5).

PARROTT, J. AND MACKENZIE, N., 2000. *Restoring and managing riparian woodlands*. Aberfeldy, Perthshire: Scottish Native Woods.

PARSONS, H. AND GILVEAR, D.J., 2002. Valley floor landscape change following almost 100 years of flood embankment abandonment on a naturally wandering gravel bed river. *River Research and Applications*, 18 (5), 461-479.

PATTISON, I. AND LANE, S.N., 2012. The link between land-use management and fluvial flood risk: A chaotic conception? *Progress in Physical Geography*, 36 (1), 72-92.

PATTISON, I.S., LANE, S.N., HARDY, R.J. AND REANEY, S.M., 2014. The role of tributary relative timing and sequencing in controlling large floods. *Water Resources Research*, 50 (7), 5444-5458.

PATTO, P.M., CLEMENT, C.R. AND FORBES, T.J., 1978. Grassland poaching in England and Wales. Maidenhead: Grassland Research Institute.

PENNING-ROWSELL, E.C, JOHNSON, C., TUNSTALL, S., TAPSELL, S., MORRIS, J., CHATTERTON, J.B AND GREEN, C., 2005. *The Benefits of Flood and Coastal Risk Management: A Manual of Assessment Techniques*. Multi-Coloured Manual. London: Middlesex University Press [revised 2011, 2013].

PETIT, S., STUART, R.C., GILLESPIE, M.K. AND BARR, C.J., 2001. Woody field boundaries in Great Britain: stock and change between 1990 and 1998. In *Hedgerows of the World: the ecological functions in different landscapes*. Proceedings of the 10th annual conference of the International Association for Landscape Ecology (Birmingham, 2001), pp. 111-116. International Association for Landscape Ecology UK.

PETIT, S., STUART, R.C., GILLESPIE, M.K. AND BARR, C.J., 2003. Field boundaries in Great Britain: stock and change between 1984, 1990 and 1998. *Journal of Environmental Management*, 67 (3), 229-238.

PETHICK, J., 1995. *An Introduction to Coastal Geomorphology*. London: Edward Arnold.

PETTS, G.E. AND FOSTER, I.D.L., 1985. Channel morphology. In *Rivers and Landscape* (ed. G. Petts and I. Foster). pp. 140-174. London: Edward Arnold.

PEVENSEY COASTAL DEFENCE, 2015. *Reuse of sand from Sovereign Harbour maintenance dredging*. Hailsham, Sussex: Wealden District Council. Available from: <u>http://www.pevenseyparishcouncil.gov.uk/pdf/d.councilMinReports.pdf</u> [Accessed 7 July 2017]. PEVENSEY COASTAL DEFENCE, undated. *Beach maintenance* [online]. Fareham, Hampshire: Pevensey Coastal Defence Ltd. Available from: <u>http://www.pevensey-</u> <u>bay.co.uk/beach-maintenance.html</u> [Accessed 7 July 2017].

PIÉGAY, H. AND BRAVARD J.P., 1996. Response of a Mediterranean riparian forest to a 1 in 400 year flood, Ouveze River, Drome-Vaucluse, France. *Earth Surface Processes and Landforms*, 22 (1), 31-43.

PIÉGAY, H., DARBY, S.E., MOSSELMAN, E. AND SURIAN, N., 2005. A review of techniques available for delimiting the erodible river corridor: A sustainable approach to managing bank erosion. *River Research and Applications*, 21 (7), 773-789.

PIERZYNSKI, G.M., SIMS, J.T. AND VANCE, G.F., 2000. Soil and Environmental *Quality*, 2nd edition. Boca Raton, FL: CRC Press.

PILKINGTON, M., WALKER, J., MASKILL, R., ALLOTT, T. AND EVANS, M., 2015. *Restoration of blanket bogs; flood risk reduction and other ecosystem benefits*. Final report of the Making Space for Water project. Edale, Derbyshire: Moors for the Future Partnership. Available from:

http://www.moorsforthefuture.org.uk/default/files/Summary.a.pdf [Accessed 7 July 2017].

PINAY, G., GUMIERO, B., TABACCHI, E., GIMENEZ, O., TABACCHI-PLANTY, A.M., HEFTING, M.M., BURT, T.P., BLACK, V.A., NILSSON, C. AND IORDACHE, V., 2007. Patterns of denitrification rates in European alluvial soils under various hydrological regimes. *Freshwater Biology*, 52 (2), 252-266.

PINTER, N., VAN DER PLOEG, R.R., SCHWEIGERT, P. AND HOEFER, G., 2006. Flood magnification on the River Rhine. *Hydrological Processes*, 20 (1), 147-164.

PITT, M., 2008. *The Pitt Review: lessons learned from the 2007 floods.* London: Cabinet Office.

PLANCHON, O., ESTEVES, M., SILVERA, N. AND LAPETITE, J.M., 2001. Microrelief induced by tillage: measurement and modeling of surface storage capacity. *Catena*, 46 (2-3), 141-157.

PLYMOUTH COASTAL OBSERVATORY, 2014. *Review of South West Coast beach response to wave conditions during the winter of 2013-2014*. South West Regional Coastal Monitoring Programme. Teignbridge, Devon: Plymouth Coastal Observatory.

PONTEE, N.I., 2015. Impact of managed realignment design on estuarine water levels. *Proceedings of the Institution of Civil Engineers - Maritime Engineering*, 168 (2), 48-61.

PONTEE, N.I., NARAYAN, S., BECK, M., HOSKING A.H., 2016. Building with nature: lessons from around the world. *Maritime Engineering Journal*, 169 (1), 29-36.

POLLOCK, M.M., BEECHIE, T.J., WHEATON, J.M., JORDAN, C.E., BOUWES, N., WEBER, N. and VOLK, C. 2014. Using beaver dams to restore incised stream ecosystems. *BioScience*, 64 (4), 279-290.

POLVI, L.E. and WOHL, E., 2012. The beaver meadow complex revisited – the role of beavers in post-glacial floodplain development. *Earth Surface Processes and Landforms*, 37 (3), 332-346.

POOLE BOROUGH COUNCIL, 2016. *Poole Bay Nearshore Replenishment Trial, November 2014 to April 2016. Monitoring report.* Poole: Poole Borough Council.

POST, 2015. *Ecosystem service valuation*. POST-PN-378. London: Parliamentary Office for Science and Technology.

POSTHUMUS, H. AND MORRIS, J., 2010. Implications of CAP reform for land management and runoff control in England and Wales. *Land Use Policy*, 27 (1), 42-50.

POSTHUMUS, H., HEWETT, C.J.M., MORRIS, J. AND QUINN, P.F., 2008. Agricultural land use and flood risk management: engaging with stakeholders in north Yorkshire. *Agricultural Water Management*, 95 (7), 787-798.

PREMOV, A., COXON, C. E., HACKETT, R., KIRWAN, L. AND RICHARDS, K. G., 2014. Effects of over-winter green cover on soil solution nitrate concentrations beneath tillage land. *Science of the Total Environment*, 470, 967-974.

PRITCHETT, W.L. AND FISHER, R.F., 1987. *Properties and Management of Forest Soils*, 2nd edition. New York: John Wiley and Sons

PUTTOCK, A. AND BRAZIER, R., 2014. Culm grasslands proof of concept phase 1: developing an understanding of the hydrology, water quality and soil resources of unimproved grasslands. Exeter: Devon Wildlife Trust.

PUTTOCK, A., GRAHAM, H.A., CUNLIFFE, A.M., ELLIOTT, M. AND BRAZIER, R.E., 2017. Eurasian beaver activity increases water storage, attenuates flow and mitigates diffuse pollution from intensively-managed grasslands. *Science of the Total Environment*, 576, 430-443.

PYE, K. AND BLOTT, S.J., 2012. A geomorphological survey of Welsh dune systems to determine best methods of dune rejuvenation. CCW Science Report No. 1002. Bangor: Countryside Council for Wales.

PYE, K., SAYE, S. AND BLOTT, S., 2007. *Management for flood and coastal defence, Part 4: Techniques for sand dune management.* Defra/Environment Agency R&D Technical Report FD1392/TR. London: Department for Environment, Food and Rural Affairs.

PYE, K., BLOTT, S.J. AND HOWE, M.A., 2014. Coastal dune stabilization in Wales and requirements for rejuvenation. *Journal of Coastal Conservation*, 18 (1), 27-54.

QIU, Z., 2003. VSA-based strategy for placing conservation buffers in agricultural watersheds. *Environmental Management*, 32 (3), 299-311.

QUINN, P., 2016a. A natural engineering approach to flood management. Presentation to Upland Hydrology Conference. Available from: <u>http://www.uplandhydrology.org.uk/wp-content/uploads/2016/01/Paul-Quinn.pdf</u> [Accessed 13 July 2017].

QUINN, P., 2016b. *Holding water: working with nature to ease floods and droughts*. Policy Note 2. Newcastle upon Tyne: Institute of Sustainability, Newcastle University.

QUINN, P.F., HEWETT, C.J.M., JONCZYK, J., AND GLENIS, V.A., 2007a. *The PROACTIVE approach to Farm Integrated Runoff Management (FIRM) plans: flood storage on farms*. Newcastle upon Tyne: Newcastle University. Available from: <u>https://research.ncl.ac.uk/proactive/ms4w/PROACTIVEReportPQ.pdf</u> [Accessed 7 July 2017].

QUINN, P., JONCZYK, J., RIMMER, F. AND HEWETT, C., 2007b. 'Store, slow and filter'. The PROACTIVE approach to Farm Integrated Runoff Management (FIRM) plans with respect to nutrients. Newcastle upon Tyne: Newcastle University. Available from: <u>https://research.ncl.ac.uk/proactive/ms4w/Proactivenutrients.pdf</u> [Accessed 6 July 2017].

QUINN, P., BURKE, S., JONCZYK, J., HEWITT, C., WILKINSON, M. and RIMMER, D., 2008a. *Flood storage and attenuation on farms. Making Space for Water (MS4W): Nafferton Farm water quantity study.* Newcastle upon Tyne: Newcastle University.

Available from: <u>http://research.ncl.ac.uk/proactive/ms4w/firmflood%20leaflet.pdf</u> [Accessed 6 July 2017].

QUINN, P., BURKE, S., JONCZYK, J., HEWITT, C., WILKINSON, M. AND RIMMER, D. 2008b. *Nutrient trapping and recycling on farms. Making Space for Water (MS4W): Water quality study.* Newcastle upon Tyne: Newcastle University. Available from: <u>http://research.ncl.ac.uk/proactive/ms4w/firmnutrient%20leaflet.pdf</u> [Accessed 6 July 2017].

QUINN, P., BURKE, S., JONCZYK, J., HEWITT, C., WILKINSON, M. AND RIMMER, D. 2008c. *Farm Integrated Runoff Management Plans. Flood storage and attenuation on farms.* Newcastle upon Tyne: Newcastle University. Available from <u>https://research.ncl.ac.uk/proactive/FIRMandFRMversion3.pdf</u> [Accessed 19 July 2017].

QUINN, P., O'DONNELL, G., NICHOLSON, A., WILKINSON, M., OWEN, G., JONCZYK, J., BARBER, N., HARDWICK N. AND DAVIES, G., 2013. *Potential use of runoff attenuation features in small rural catchments for flood mitigation: evidence from Belford, Powburn and Hepscott.* Joint Newcastle University, Royal Haskoning and Environment Agency report. Available from:

https://research.ncl.ac.uk/proactive/belford/newcastlenfmrafreport/reportpdf/June%20N FM%20RAF%20Report.pdf [Accessed 7 July 2017].

RAHMAN, M.M., THOMPSON, J.R. AND FLOWER, R.J., 2016. An enhanced SWAT wetland module to quantify hydraulic interactions between riparian depressional wetlands, rivers and aquifers. *Environmental Modelling and Software*, 84, 263-289.

RAK, G., KOZELJ, D. and STEINMAN, F., 2016. The impact of floodplain land use on flood wave propagation. *Natural Hazards*, 83 (1), 425-443.

RALLINSON, R.E., 1980. Origin and evolution of the SCS runoff equation. In *Proceedings of Symposium on Watershed Management* (Boise, ID, 1980), pp. 912-924. New York: American Society of Civil Engineers.

RAMCHUNDER, S.J., BROWN, L.E. AND HOLDEN, J., 2009. Environmental effects of drainage, drain-blocking and prescribed vegetation burning in UK upland peatlands. *Progress in Physical Geography*, 33 (1), pp. 49-79.

RAMCHUNDER, S. J., BROWN, L. E. AND HOLDEN, J. (2012). Catchment-scale peatland restoration benefits stream ecosystem biodiversity. *Journal of Applied Eco*logy, 49(1), 182-191.

RAMESHWARAN, P. AND SHIONO, K., 2007. Quasi two-dimensional model for straight overbank flows through emergent. *Journal of Hydraulic Research*, 45 (3), 302-315.

RAMESHWARAN, P. AND NADEN, P., 2012. Modelling the influence of macrophyte patches on river flow. In *River Flow 2012* (ed. R.E. Murillo Muñoz), pp. 163-170. Boca Raton, FL: CRC Press.

RAMESHWARAN, P., NADEN, P.S. AND LAWLESS, M., 2011. Flow modelling in gravel-bed rivers: rethinking the bottom boundary condition. *Earth Surface Processes and Landforms*, 36 (10), 1350-1366.

RAMESHWARAN, P., SUTCLIFFE, A., NADEN, P. AND WHARTON, G., 2014. Modelling river flow responses to weed management. In *River Flow 2014* (ed. A.J. Schleiss, G. de Cesare and M. Franca), pp. 467-474. Boca Raton, FL: CRC Press.

RANDALL, R.E., 2004. Management of coastal vegetated shingle in the United Kingdom. *Journal of Coastal Vegetation*, 10 (1-2), 159-168.

RANZI, R., BOCHICCHIO, M., AND BACCHI, B., 2002. Effects on floods of recent afforestation and urbanisation in the Mella River (Italian Alps). *Hydrology and Earth System Sciences Discussions*, 6 (2), 239-254.

RAVEN, E.K., LANE, S.N., FERGUSON, R.I. AND BRACKEN, L.J., 2009. The spatial and temporal patterns of aggradation in a temperate, upland, gravel-bed river. *Earth Surface Processes and Landforms*, 34 (9), 1181-1197.

REANEY, S. AND PEARSON, C., 2014. Spatial targeting of natural flood risk management within large river catchments: a nested approach of SCIMAP-Flood and CRUM3. Durham: Durham University. Available from: <u>https://consult.defra.gov.uk/water-and-flood-risk-management/flood-risk-management-modelling-competition/results/12 reaney and pearson_river_eden_scimap_flood_and_crum3.pdf [Accessed 7 July 2017].</u>

REANEY, S.M., BRACKEN, L.J. AND KIRKBY, M.J., 2007. Use of the connectivity of runoff model (CRUM) to investigate the influence of storm characteristics on runoff generation and connectivity in semi-arid areas. *Hydrological Processes*, 21 (7), 894-906.

REEVES, D.W., NORFLEET, M. L., ABRAHAMSON, D.A., SCHOMBERG, H.H., CAUSARANO, H. AND HAWKINS, G.L., 2005. Conservation tillage in Georgia: economics and water resources. In *Proceedings of the 2005 Georgia Water Resources Conference* (Georgia, 2005), edited by K.J. Hatcher, pp. 665-668. Athens, GA: University of Georgia, Institute of Ecology.

REID, I. AND PARKINSON, R.J., 1984. The nature of the tile-drain outfall hydrograph in heavy clay soils. *Journal of Hydrology*, 72 (3-4), 289-305.

RESOP, J., HESSION, W. and WYNN-THOMPSON, T., 2014. Quantifying the parameter uncertainty in the cross-sectional dimensions for a stream restoration design of a gravel-bed stream. *Journal of Soil and Water Conservation*, 69, 306-315.

RESTREPO, J.I., MONTOYA, A.M. AND OBEYSEKERA, J., 1998. A wetland simulation module for the MODFLOW ground water model. *Ground Water*, 36 (5), 764-770.

RIQUIER, J., PIÉGAY, H. AND ŠULC MICHALKOVÁ, M., 2015. Hydromorphological conditions in eighteen restored floodplain channels of a large river: linking patterns to processes. *Freshwater Biology*, 60 (6), 1085-1103.

RITZEMA, H.P. (ED.), 1994. *Drainage principles and applications*. ILRI Publication 16. Wageningen, The Netherlands: International Institute for Land Reclamation and Improvement.

RIVER RIBBLE TRUST, 2015. *Catchment Restoration Fund (CRF): Limestone Ribble project area. Outcomes and achievements.* Clitheroe, Lancashire: River Ribble Trust.

ROBERTS, J.M., 1983. Forest transpiration: a conservative hydrological process? *Journal of Hydrology*, 66, 133-141.

ROBERTS, J.M., ROSIER, P.T.W. AND SMITH, D.M., 2001a. *Effects of afforestation on chalk groundwater resources.* Centre for Ecology and Hydrology report to Defra. Wallingford: Centre for Ecology and Hydrology.

ROBERTS, S., VERTESSY, R., AND GRAYSON, R., 2001b. Transpiration from *Eucalyptus sieberi* (L. Johnson) forests of different age. *Forest Ecology and Management*, 143 (1-3), 153-161.

ROBINSON, M., 1990. *Impact of improved land drainage on river flows.* Report No. 113. Wallingford: Institute of Hydrology.

ROBINSON, M. AND ARMSTRONG, A.C., 1988. The extent of agricultural field drainage in England and Wales, 1971–1980. *Transactions of the Institute of British Geographers*, 13, 19-28.

ROBINSON, M. AND DUPEYRAT, A., 2005. Effects of commercial timber harvesting on streamflow regimes in the Plynlimon catchments, mid-Wales. *Hydrological Processes*, 19 (6), 1213-1226.

ROBINSON, M., RYDER, E.L. AND WARD, R.C., 1985. Influence on streamflow of field drainage in a small agricultural catchment. *Agricultural Water Management*, 10 (2), 145-158

ROBINSON, M., MOORE, R.E., NISBET, T.R. AND BLACKIE, J.R., 1998. *From moorland to forest: the Coalburn catchment experiment.* Institute of Hydrology Report No. 133. Wallingford: Centre for Ecology and Hydrology.

ROBINSON, M., COGNARD-PLANCQ, A,-L., COSANDEY, C., DAVID, J., DURAND, P., FÜHRER, H,-W., HALL, R., HENDRIQUES, M.O., MARC, V., MCCARTHY, R., MCDONNELL, M., MARTIN, C., NISBET, T., O'DEA, P., RODGERS, M. AND ZOLLNER, A., 2003. Studies of the impact of forests on peak flows and baseflows: a European perspective. *Forest Ecology and Management*, 186 (1), 85-97.

ROGGER, M., AGNOLETTI, M., ALAOUI, A., BATHURST, J.C., BODNER, G., BORGA, M., CHAPLOT, V., GALLART, F., GLATZEL, G., HALL, J., HOLDEN, J., HOLOI, L., HORN, R., KISS, A., KOHNOVA, S., LEITINGER, G., LENMARTZ, B., PARAJKA, J., PERDIGAO, R., PETH, S., PLAVCOVA, L., QUINTON, QUINTON, J.N., ROBINSON, M., SALINAS, J.L., SANTORO, A., SZOLGAY, J., TRON, S., VAN DEN AKKER, J.J.H., CAN DEN, VIGLIONE., A. AND BLÖSCH, G., 2017. Land-use change impacts on flood at the catchment scale: challenges and opportunities for future research. *Water Resources Research*, DOI: 10.1002/2017WR020723.

ROHDE, S., HOSTMANN, M., PETER, A. & EWALD, K. C. 2006. Room for rivers: An integrative search strategy for floodplain restoration. *Landscape and Urban Planning*, 78, 50-70.

RONGRONG, W. AND GUISHAN, Y., (2007) Influence of land use/cover change on storm runoff. A case study of Xitiaoxi River Basin in upstream of Taihu Lake Watershed. *Chinese Geographical Science*, 17 (4), 349-356.

RONI, P. and QUINN, T.P., 2001. Effects of wood placement on movements of trout and juvenile coho salmon in natural and artificial stream channels. *Transactions of the American Fisheries Society*, 130 (4), 675-685.

RONI, P. AND BEECHIE, T. (ED.), 2012. Stream and Watershed Restoration: A Guide to Restoring Riverine Processes and Habitats. Oxford: Wiley-Blackwell.

ROTHACHER, J., 1963. Net precipitation under a Douglas-fir forest. *Forest Science*, 9 (4), 423-429.

ROWLANDS, O.E., 2011. *Current trends in the costs of managed realignment and regulated tidal exchange schemes in the UK*. MSc thesis, University of Southampton.

ROYAL HASKONINGDHV, 2015a. *North Norfolk sandscaping: feasibility study.* Report for the Crown Estate. Peterborough: Royal Haskoning DHV UK.

ROYAL HASKONINGDHV, 2015b. *Sandscaping high potential sites: England*. Peterborough: Royal Haskoning DHV UK.

RPA, RHDDV AND ALLATHAN ASSOCIATES, 2015. *Project RPA/001/14: Assessing the mechanisms for compensating land managers.* Report prepared for the Scottish Government. Loddon, Norfolk: Risk & Policy Analysts.

RRC, 2012. *Practical River Restoration Appraisal Guidance for Monitoring Options (PRAGMO).* Cranfield, Bedfordshire: River Restoration Centre.

RRC, 2016. *Manual of river restoration techniques* [online]. Cranfield, Bedfordshire: River Restoration Centre. Available from: <u>http://www.therrc.co.uk/manual-river-</u> <u>restoration-techniques</u> [Accessed 7 July 2017].

RSPB, 2005. The saltmarsh creation handbook: a project manager's guide to the creation of saltmarsh and intertidal mudflat. Sandy, Bedfordshire: RSPB.

RSPB, 2006. Farming for wildlife in Wales: hedgerow management. Bangor: RSPB.

RSPB, 2008a. *Farm hedges and their management*. Information leaflet. Sandy, Bedfordshire: RSPB.

RSPB, 2008b. Working with natural processes to reduce flood risk: an RSPB perspective. Sandy, Bedfordshire: RSPB.

RUIZ-VILLANUEVA, V., BODOQUE, J.M., DIEZ-HERRERO, A., EGUIBAR, M.A. AND PARDO-IGUZQUIZA, E., 2013. Reconstruction of a flash flood with large wood transport and its influence on hazard patterns in an ungauged mountain basin. *Hydrological Processes*, 27 (24), 3424-3437.

RUIZ-VILLANUEVA, V., PIÉGAY, H., GURNELL, A.A., MARSTON, R.A. AND STOFFEL, M., 2016. Recent advances quantifying the large wood dynamics in river basins: New methods and remaining challenges. *Reviews of Geophysics*, 54 (3), 611-652.

RUPP-ARMSTRONG, S. AND NICHOLLS, R.J., 2007. Coastal and estuarine retreat: a comparison of the application of managed realignment in England and Germany. *Journal of Coastal Research*, 23 (6), 1418-1430.

SALO, J., KALLIOLA, R., HAKKINEN, I., MAKINEN, Y., NIEMELA, P., PUHAKKA, M. AND COLEY, P.D., 1986. River dynamics and the diversity of Amazon lowland forest. *Nature*, 322, 254-258.

SAMSOM, A.L., 1999. Upland vegetation management: the impacts of overstocking. *Water Science and Technology*, 39 (12), 85-92.

SATHER, J.H. AND SMITH, R.D., 1972. *An overview of major wetland functions and values*. FWS/OBS-84/18. Washington DC: US Fish and Wildlife Services, Department of Interior.

SCHAFER, K., 1986. Experiments with catch crops for erosion control in maize production. *Wirtschaftseigene Futter*, 32, 60-71.

SCHOLEFIELD, D. AND HALL, D.M., 1986. A recording penetrometer to measure the strength of soil in relation to the stresses exerted by a walking cow. European Journal of Soil Science, 37 (1), 165-176.

SCHWAB, G.O., FANGMEIER, D.D., FREVERT, R.K. AND ELLIOT, W.J., 1993. *Soil and Water Conservation Engineering*, 4th edition. New York: John Wiley and Sons.

SEAR, D. AND NEWSON, M., 2004. The hydraulic impact and performance of a lowland rehabilitation scheme based on pool–riffle installation: the River Waveney, Scole, Suffolk, UK. *River Research and Applications*, 20 (7), 847-863.

SEAR, D., WILCOCK, D., ROBINSON, M. AND FISHER, K., 2000. River channel modification in the UK. In *The Hydrology of the United Kingdom: a Study of Change* (ed. M. Acreman). Oxford: Routledge.

SEAR, D., KITTS, D. AND MILLIGAN, C., 2006. New Forest LIFE-III monitoring report: the geomorphic and hydrological response of New Forest streams to river restoration. Southampton: University of Southampton.

SEAR, D.A., MILLINGTON, C.E., KITTS, D.R. AND JEFFRIES, R., 2010. Logjam controls on channel:floodplain interactions in wooded catchments and their role in the formation of multi-channel patterns. *Geomorphology*, 116 (3-4), 305-319.

SEFTON COUNCIL, 2011. *Hightown Dune Restoration Project. Project Appraisal Report.* Bootle, Merseyside: Sefton Metropolitan Borough Council.

SEFTON COUNCIL, 2012. Hightown Dune Restoration Project, Update – November 2012 (PowerPoint presentation by Graham Lymbery of Sefton Council).

SENDZIMIR, J., DUBEL, A., LINNEROOTH-BAYER, J., DAMURSKI, J. AND SCHROETER, D., 2014. Assessment of the costs, risks and benefits of selected integrated policy options to adapt to flood and drought in the water and agricultural sectors of the Warta River Basin. EGU General Assembly Conference Abstracts 2014, abstract 3679.

SENTER, A.E. and PASTERNACK, G.B., 2011. Large wood aids spawning Chinook salmon (*Oncorhynchus Tshawytscha*) in marginal habitat on a regulated river in California. *River Research and Applications*, 27 (5), 550-565.

SEPA, 2015. *Natural Flood Management Handbook.* Stirling: Scottish Environment Protection Agency.

SEPA AND FORESTRY COMMISSSION SCOTLAND, 2012. Flood Risk Management (Scotland) Act 2009: methods to screen and quantify natural flood management effects. Stirling: Scottish Environment Protection Agency.

SHAW, E.M. 1983. *Hydrology in Practice*. Van Nostrand Reinhold, Wokingham, England.

SHEAFFER, J.R., MULLAN, J.D. and HINCH, N.B., 2002. Encouraging wise use of floodplains with market-based incentives. *Environment: Science and Policy for Sustainable Development*, 44 (1), 32-43.

SHEPHERD, D, BURGESS, D., JICKELLS, T., ANDREWS, J., CAVE, R., TURNER, R.K., ALDRIDGE, J., PARKER, E.R. AND YOUNG, E., 2007. Modelling the effects and economics of managed realignment on the cycling and storage of nutrients, carbon and sediments in the Blackwater estuary, UK. *Estuarine, Coastal and Shelf Science*, 73 (3-4), 355-367.

SHEPHERD, M., LABADZ, J., CAPORN, S., CROWLE, A., GOODISON, R., REBANE, M. AND WATERS, R., 2013. *Restoration of blanket bog.* Report NEER003. York: Natural England.

SHIELDS, F.D. AND GIPPEL, C.J., 1995. Prediction of effects of woody debris removal on flow resistance. *Journal of Hydraulic Engineering*, 121 (4), 341-354.

SHIELDS, F.D., JR., COOPER, C.M., KNIGHT, S.S. AND MOORE, M.T., 2003. Stream corridor restoration research: a long and winding road. *Ecological Engineering*, 20, 441-454.

SHOLTES, J.S. AND DOYLE, M.W. 2011. Effect of channel restoration on flood wave attenuation. *Journal of Hydraulic Engineering*, 137 (2), 196-208.

SHORE, M., JORDAN, P., MELLANDER, P.-E., KELLY-QUINN, M. AND MELLAND, A.R., 2015. An agricultural drainage channel classification system for phosphorus management. *Agriculture, Ecosystems and Environment,* 199, 207-215.

SIBBENSEN, E., HANSEN, A.C., NIELSEN, J.D. AND HEIDMAN, R., 1994. Runoff, erosion and phosphorus loss from various cropping systems in Denmark. In *Conserving Soil Resources: European Perspectives* (ed. R.J. Rickson), pp. 87-93. Wallingford: CABI Publishing.

SIMON, A., 1989. A model of channel response in disturbed alluvial channels. *Earth Surface Processes and Landforms*, 14, 11-26.

SIMON, A. and RINALDI, M., 2006. Disturbance, stream incision, and channel evolution: The roles of excess transport capacity and boundary materials in controlling channel response. *Geomorphology*, 79, 361-383.

SKELCHER, G., 2008. *Fylde Sand Dunes Management Action Plan*. Lytham St Annes, Lancashire: Fylde Borough Council.

SLATER, L. J., SINGER, M. B. & KIRCHNER, J. W. 2015. Hydrologic versus geomorphic drivers of trends in flood hazard. *Geophysical Research Letters*, 42, 370-376.

SMITH, K.A., 2012. *Evaluation of land management impacts on low flows in northern England*. Master's thesis, Durham University. Available from: http://etheses.dur.ac.uk/3501/ [Accessed 7 July 2017].

SOANE, B.D., BALL, B.C., ARVIDSSON, J., BASCH, G., MORENO, F. AND ROGER-ESTRADE, J., 2012. No-till in northern, western and south-western Europe: a review of problems and opportunities for crop production and the environment. *Soil and Tillage Research*, 118, 66-87.

SOLARI, L., VAN OORSCHOT, M., BELLETTI, B., HENDRIKS, D., RINALDI, M. AND VARGAS-LUNA, A., 2016. Advances on modelling riparian vegetation – hydromorphology interactions. *River Research and Applications*, 32 (2), 164-178.

SOLLOWAY, I., 2012. *Effects of tree shelterbelts on the hydrology of upland areas.* PhD thesis, Imperial College London.

SMITH, B., CLIFFORD, N.J. AND MANT, J., 2014. The changing nature of river restoration. *WIREs Water* 1 (3), 249-261.

SMITHERS, R., KORKEALA, O., BRACE, S., WHITELEY, G. AND HOLMES, B., 2016. *Valuing flood-regulation services for inclusion in the UK ecosystem accounts*. Report prepared for the UK Office for National Statistics. Harwell, Didcot: Ricardo Energy and Environment.

SPEIRS, R.B. AND FROST, C.A., 1985. The increasing incidence of accelerated soil water erosion on arable land in the east of Scotland. *Research and Development in Agriculture*, 2, 161-168.

SPRAY, C.J. (ED.), 2017. *Eddleston Water: Project Report 2016*. Melrose, Roxburghshire: Tweed Forum. Available from: <u>http://tweedforum.org/publications/pdf/Eddleston_Report_Jan_2017.pdf</u> [Accessed 7 July 2017].

SRINIVASAN, M.S. AND MCDOWELL, R.W., 2009. Identifying critical source areas for water quality: 1. Mapping and validating transport areas in three headwater catchments in Otago, New Zealand. *Journal of Hydrology*, 379 (1), 54-67.

STEDNICK, J.D., 1996. Monitoring the effects of timber harvest on annual water yield. *Journal of Hydrology*, 176 (1-4), 79-95.

STEWART, A.J.A. AND LANCE, A.N., 1983. Moor-draining: a review of impacts on land use. *Journal of Environmental Management*, 17 (1), 81-99.

STEWART, G.B., BAYLISS, H.R., SHOWLER, D.A., PULLIN, A.S. AND SUTHERLAND, W.J., 2006. *Does the use of in-stream structures and woody debris increase the abundance of salmonids?* Systematic Review No. 12. Bangor: Centre for Evidence-Based Conservation.

STOCKAN, J. A., BAIRD, J., LANGAN, S. J., YOUNG, M. R. and IASON, G. R., 2014. Effects of riparian buffer strips on ground beetles (Coleoptera, Carabidae) within an agricultural landscape. *Insect conservation and diversity*, **7**, 172-184.

STORCK, P., BOWLING, L.C., WETHERBEE, P. AND LETTENWAIER, D., 1998. Application of a GIS-based distributed hydrology model for prediction of forest harvest effects on peak stream flow in the Pacific Northwest. *Hydrological Processes*, 12 (6), 889-904.

STRATFORD, C., BREWIN, P., ACREMAN, M. AND MOUNTFORD, O., 2015. A simple model to quantify the potential trade-off between water level management for ecological benefit and flood risk. *Ecohydrology and Hydrobiology*, 15 (3), 150-159.

STRUDLEY, M.W., GREEN, T.R. AND ASCOUGH, J.C., 2008. Tillage effects on soil hydraulic properties in space and time: state of the science. *Soil and Tillage Research*, 99 (1), 4-48.

STUDDERT, G.A. AND ECHEVERR, H.E., 2000. Crop rotations and nitrogen fertilization to manage soil organic carbon dynamics. *Soil Science Society of America Journal*, 64 (4), 1496-1503.

STUTTER, M.I., CHARDON, W.J. AND KRONVANG, B., 2012. Riparian buffer strips as a multifunctional management tool in agricultural landscapes: introduction. *Journal of Environmental Quality*, 41 (2), 297-303.

SUN, G., RIEKERK, H. AND COMERFORD, N.B., 1998. Modeling the hydrologic impacts of forest harvesting on Florida flatwoods. *Journal of the American Water Resources Association*, 34 (4), 843-854.

SUN, X., SHIONO, K., RAMESHWARAN, P. AND CHANDLER, J.H., 2010. Modelling vegetation effects in irregular meandering river. *Journal of Hydraulic Research*, 48 (6), 775-783.

SURIAN, N., ZILIANI, L., COMITI, F., LENZI, M.A. AND MAO, L., 2009. Channel adjustments and alteration of sediment fluxes in gravel-bed rivers of north-eastern Italy: potentials and limitations for channel recovery. *River Research and Applications*, 25 (5), 551-567.

SUSSEX FLOW INITIATIVE, 2016. *Natural flood management guidance: woody dams, deflectors and diverters*. Grantham: Woodland Trust.

ŚWIĄTEK, D., SZPORAK, S., CHORMAŃSKI, J. AND OKRUSZKO, T., 2008. Hydrodynamic model of the Lower Biebrza River flow – a tool for assessing the hydrologic vulnerability of a floodplain to management practices. *Ecohydrology and Hydrobiology*, 8 (2-4), 331-337.

TEKLEHAIMANOT, Z., JARVIS, P.G. AND LEDGER, D.C., 1991. Rainfall interception and boundary layer conductance in relation to tree spacing. *Journal of Hydrology*, 123 (3-4), 261-278.

THE FLOW PARTNERSHIP, 2015. 5% Future. Holding water in the landscape: a new practical deal for farmers and land managers [online]. Available from: http://research.ncl.ac.uk/proactive/5future/ [Accessed 10 July 2017]

THE RIVERS TRUST, 2014. *Managing ditches – practical examples.* PINPOINT Best Practice Information Sheet. Soil Management Sheet 21.0b. Callington, Cornwall: The
Rivers Trust. Available from: <u>http://www.theriverstrust.org/media/2017/04/Pinpoint-21.0-Soil-Management-Managing-ditches.pdf</u> [Accessed 7 July 2017].

THOMAS, H. AND NISBET, T.R., 2006. An assessment of the impact of floodplain woodland on flood flows. *Water and Environment Journal*, 21 (2), 114-126.

THOMAS, H. AND NISBET, T., 2012. Modelling the hydraulic impact of reintroducing large woody debris into watercourses. *Journal of Flood Risk Management*, 5 (2), 164-174.

THOMAS, H. AND NISBET, T.R., 2016. Slowing the Flow at Pickering: quantifying the effect of catchment woodland planting on flooding using the Soil Conservation Service Curve Number method. *International Journal of Safety and Security Engineering*, 6 (3), 466-474.

THOMPSON, C., FRYIRS, K. and CROKE, J., 2015. The disconnected sediment conveyor belt: patterns of longitudinal and lateral erosion and deposition during a catastrophic flood in the Lockyer Valley, South East Queensland, Australia. *River Research and Applications*.

THORNE, C. R., HEY, R. D. and NEWSON, M. D., 1997. *Applied fluvial geomorphology for river engineering and management*, John Wiley & Sons Ltd.

THORNE, C.R., LAWSON, E.C., OZAWA, C., HAMLIN, S.L. AND SMITH, S.A., 2015. Overcoming uncertainty and barriers to adoption of Blue-Green Infrastructure for urban flood risk management. *Journal of Flood Risk Management*, DOI: 10.1111/jfr3.12218.

TINCH, R. AND LEDOUX, L., 2006. *Economics of managed realignment in the UK. Final report to the Coastal Futures Project.* Report prepared by Environmental Futures Ltd report to the RSPB.

TOCKNER, K. AND STANFORD, J.A. 2002. Riverine flood plains: present state and future trends. *Environmental Conservation*, 29 (3), 308-330.

TOWNEND, I. AND PETHICK, J., 2002, Estuarine flooding and managed retreat. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, 360 (1796), 1477-1495.

TSUKAMOTO, Y. 1975. Effect of forest litter on runoff cycle in a small experimental watershed. In: Proceedings of Symposium on Hydrological Characteristics of River Basins. Tokyo, Japan, Dec. 1975. *International Association of Hydrological Sciences Publication* 117, 487-495.

TULLBERG, J.N., YULE, D.F. AND MCGARRY, D., 2007. Controlled traffic farming – from research to adoption in Australia. *Soil and Tillage Research*, 97 (2), 272-281.

UK NATIONAL ECOSYSTEM SERVICES ASSESSMENT, 2014. *The UK National Ecosystem Assessment: synthesis of the key findings.* Cambridge: UN Environment Programme World Conservation Monitoring Centre.

VAAJA, M., HYYPPÄ, J., KUKKO, A., KAARTINEN, H., HYYPPÄ, H. AND ALHO, P., 2011. Mapping topography changes and elevation accuracies using a mobile laser scanner. *Remote Sensing*, 3 (3), 587-600.

VALVERDE, R.S., 2013. *Roughness and geometry effects of engineered log jams on 1-D flow characteristics*. Master's thesis, Oregon State University.

VAN DEN BELT, M. AND COSTANZA, R. (ED.), 2011. 12: Ecological Economics of Estuaries and Coasts. Volume 12 of 12-part series, *Treatise on Estuarine and Coastal Science* (ed. D.S. McLusky and E. Wolanski). London: Academic Press. VAN HULZEN, J.B., SEGERS, R., VAN BODEGOM, P.M. AND LEFFELAAR, P.A. 1999. Temperature effects on soil methane production: an explanation for observed variability. *Soil Biology and Biochemistry*, 31 (14), 1919-1929.

VÁSQUEZ-MÉNDEZ, R., VENTURA-RAMOS, E., OLESCHKO, K., HERNÁNDEZ-SANDOVAL, L., PARROT, J.F. AND NEARING, M.A., 2010. Soil erosion and runoff in different vegetation patches from semiarid Central Mexico. *Catena*, 80 (3), 162-169.

VERMAAT, J. E., WAGTENDONK, A. J., BROUWER, R., SHEREMET, O., ANSINK, E., BROCKHOFF, T., PLUG, M., HELLSTEN, S., AROVIITA, J. and TYLEC, L., 2016. Assessing the societal benefits of river restoration using the ecosystem services approach. *Hydrobiologia*, 769, 121-135.

VERSTRAETEN, G. AND POESEN, J., 1999. The nature of small-scale flooding, muddy floods and retention pond sedimentation in central Belgium. *Geomorphology*, 29 (3), 275-292.

VERSTRAETEN, G. AND POESEN, J., 2000. Estimating trap efficiency of small reservoirs and ponds: methods and implications for the assessment of sediment yield. *Progress in Physical Geography*, 24 (2), 219-251.

VERTESSY, R.A., WATSON, F.G.R. AND O'SULLIVAN, S.K., 2001. Factors determining relations between stand age and catchment water balance in mountain ash forests. *Forest Ecology and Management*, 143 (1-3), 13-26.

VERTESSY, R.A., ZHANG, L. AND DAWES, W.R., 2003. Plantations, river flows and salinity. *Australian Forestry*, 66 (1), 55-61.

VINTEN, A.J.A., LOADES, K., ADDY, S., RICHARDS, S., STUTTER, M., COOK, Y., WATSON, H., TAYLOR, C., ABEL, C., BAGGALEY, N., RITCHIE, R. AND JEFFREY, W., 2014. Reprint of: Assessment of the use of sediment fences for control of erosion and sediment phosphorus loss after potato harvesting on sloping land. *Science of the Total Environment*, 468, 1234-1244.

VOUGHT, L.B.M., PINAY, G., FUGLSANG, A. AND RUFFINONI, C., 1995. Structure and function of buffer strips from a water quality perspective in agricultural landscapes. *Landscape and Urban Planning*, 31 (1-3), 323-331.

VUIK, V., JONKMAN, S.N., BORSJE, B.W. AND SUZUKI, T., 2016. Nature-based flood protection: the efficiency of vegetated foreshores for reducing wave loads on coastal dikes. *Coastal Engineering*, 116, 42-56.

WAHREN, A., FEGER, K.-H., SCHWÄRZEL, K. AND MÜNCH, A., 2009. Land-use effects on flood generation – considering soil hydraulic measurements in modelling. *Advances in Geosciences*, 21, 99-107.

WAHREN, A., SCHWÄRZEL, K., AND FEGER K.-H., 2012. Potentials and limitations of natural flood retention by forested land in headwater catchments: evidence from experimental and model studies. *Journal of Flood Risk Management*, 5 (4), 321-335.

WALLING, D.E., COLLINS, A.L. AND MCMELLIN, G.K., 2003. A reconnaissance survey of the source of interstitial fine sediment recovered from salmonid spawning gravels in England and Wales. *Hydrobiologia*, 497 (1-3), 91-108.

WALTON, R., CHAPMAN, R.S. AND DAVIS, J.E. 1996. Development and application of the wetlands Dynamic Water Budget Model. *Wetlands*, 16 (3), 347-357.

WARREN, S.D., NEVILL, M.B., BLACKBURN, W.H. AND GARZA, N.E. 1986. Soil response to trampling under intensive rotation grazing. *Soil Science Society of America Journal*, 50 (5), 1336-1341.

WEIL, R. AND KREMEN, A., 2007. Thinking across and beyond disciplines to make cover crops pay. *Journal of the Science of Food and Agriculture*, 87 (4), 551-557.

WELTI, N., BONDAR-KUNZE, E., MAIR, M., BONIN, P., WANEK, W., PINAY, G. AND HEIN, T., 2012. Mimicking floodplain reconnection and disconnection using N-15 mesocosm incubations. *Biogeosciences*, 9, 4263-4278.

WENGER, S., 1999. A review of the scientific literature on riparian buffer width, extent and vegetation. Athens, GA: Office of Public Service and Outreach, Institute of Ecology, University of Georgia.

WENZEL, R., REINHARDT-IMJELA, C., SCHULTE, A. AND BOLSCHER, J., 2014. The potential of in-channel large woody debris in transforming discharge hydrographs in headwater areas (Ore Mountains, Southeastern Germany). *Ecological Engineering*, 71, 1-9.

WHARTON, G., 1995. The channel-geometry method: guidelines and applications. *Earth Surface Processes and Landforms*, 20 (7), 649-660.

WHARTON, G. AND GILVEAR, D., 2007. River restoration in the UK: meeting the dual needs of the European Union Water Framework Directive and flood defence? *International Journal of River Basin Management*, 5 (2), 143-154.

WHEATER, H. S., 2002. Progress in and prospects for fluvial flood modelling. *Philosophical Transactions of the Royal Society of London A*: Mathematical, Physical and Engineering Sciences, 360(1796), 1409-1431.

WHEATER, H.S. 2006. Flood hazard and management: a UK perspective. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, 364 (1845), 2135–2145.

WHEATER, H. AND EVANS, E., 2009. Land use, water management and future flood risk. *Land Use Policy*, 26(Suppl. 1), S251-S264.

WHEATER, H., REYNOLDS, B., MCINTYRE, N., MARSHALL, M., JACKSON, B., FROGBROOK, Z., SOLLOWAY, I., FRANCIS, O. AND CHELL, J., 2008. *Impacts of upland land management on flood risk: Multi-scale modeling methodology and results from the Pontbren experiment.* FRMRC Research Report UR16. Manchester: Flood Risk Management Research Consortium.

WHITFIELD, C.J., BAULCH, H.M., CHUN, K.P. AND WESTBROOK, C.J., 2015. Beaver-mediated methane emission: The effects of population growth in Eurasia and the Americas. Ambio, 44 (1), 7-15.

WHITTAKER, P., WILSON, C., ABERLE, J., RAUCH, H.-P. AND XAVIER, P., 2013. A drag force model to incorporate the reconfiguration of full-scale riparian trees under hydrodynamic loading. *Journal of Hydraulic Research*, 51 (5), 569-580.

WHITWORTH, G.A., 2011. *Flood exceedance attenuation: a rural solution*. MSc thesis, Aberystwyth University.

WILKINSON, M.E. AND QUINN, P.F., 2010. Belford catchment proactive flood solutions: a toolkit for managing runoff in the rural landscape. Edinburgh, Proceedings of the SAC and SEPA Biennial Conference. In *Climate, Water and Soil: Science, Policy and Practice*, Proceedings of the SAC and SEPA Biennial Conference (Edinburgh, 2010) edited by K. Crighton and R. Audsley, pp 103-110. Edinburgh: Scottish Agricultural College.

WILKINSON, M.E., QUINN, P.F., BENSON, I. AND WELTON, P., 2010a. Runoff management: mitigation measures for disconnecting flow pathways in the Belford Burn catchment to reduce flood risk. In *Managing Consequence of Changing Global*

Environment, Proceedings of the British Hydrological Society 3rd International Conference (Newcastle upon Tyne, 2010).

WILKINSON, M.E., QUINN, P.F. AND WELTON, P., 2010b. Runoff management during the September 2008 floods in the Belford catchment, Northumberland. *Journal of Flood Risk Management*, 3 (4), 285-295.

WILKINSON, M.E., HOLSTEAD, K. AND HASTINGS, E., 2013a. *Natural Flood Management in the context of UK reservoir legislation*. Aberdeen: Centre of Expertise for Waters (CREW).

WILKINSON, M.E., QUINN, P.F., BARBER, N.J. AND JONCZYK, J., 2014. A framework for managing runoff and pollution in the rural landscape using a catchment systems engineering approach. *Science of the Total Environment*, 468, 1245-1254.

WILKINSON, M.E., MACKAY, E., QUINN, P.F., STUTTER, M.I., BEVEN, K.J., MACLEOD, C.J.A., MACKLIN, M.G., ELKHATIB, Y., PERCY, B., VITOLO, C. AND HAYGARTH, P.M., 2015. A cloud based tool for knowledge exchange on local scale flood risk. *Journal of Environmental Management*, 161, 38-50.

WILLBY, N., PERFECT, C. AND LAW, A. 2014. *The Scottish Beaver Trail: Monitoring of aquatic vegetation and associated features of the Knapdale lochs 2008-2013, final report.* Scottish Natural Heritage Commissioned Report No. 688. Inverness: Scottish Natural Heritage.

WILLIAMS, L., HARRISON, S. AND O'HAGAN, A.M., 2012. *The use of wetlands for flood attenuation. Final report.* Cork, Ireland: Aquatic Services Unit, University College Cork.

WILSON, C., HOYT, J. AND SCHNAUDER, I., 2008. Impact of foliage on the drag force of vegetation in aquatic flows. *Journal of Hydraulic Engineering*, 134 (7), 885-891.

WINTERBOTTOM, S. J., 2000. Medium and short-term channel planform changes on the Rivers Tay and Tummel, Scotland. *Geomorphology*, 34, 195-208.

WITHERS, P.J.A., HODGKINSON, R.A., BATES, A. AND WITHERS, C.M., 2006. Some effects of tramlines on surface runoff, sediment and phosphorus mobilization on an erosion-prone soil. *Soil Use and Management*, 22 (3), 245-255.

WOHL, E., 2005. Compromised rivers: understanding historical human impacts on rivers in the context of restoration. *Ecology and Society*, 10, 2.

WOHL, E., LANE, S.N. AND WILCOX, A.C., 2015. The science and practice of river restoration. *Water Resources Research*, 51 (8), 5974-5997.

WOHL, E., BLEDSOE, B.P., FAUSCH, K.D., KRAMER, N., BESTGEN, K.R. AND GOOSEFF, M.N. 2016. Management of large wood in streams: an overview and proposed framework for hazard evaluation. *Journal of the American Water Resources Association*, 52 (2), 315-335.

WORRALL, F., ARMSTRONG, A. AND HOLDEN, J., 2007a. Short-term impact of peat drain-blocking on water colour, dissolved organic carbon concentration and water table depth. *Journal of Hydrology*, 337 (3-4), 315-325

WRIGHT, S., 2014. *Investigating the effectiveness of sediment traps in the Rother Valley* [online]. Arun and Western Streams Catchment Partnership. Available from: <u>http://arunwesternstreams.org.uk/projects/sediment</u> [Accessed 19 July 2017].

WRIGHT, S., 2016. Trialling in-ditch sediment traps to help limit agricultural runoff to rivers. In *Proceedings of South Downs Research Conference 2016*, pp.13-14. Available

from: <u>http://eprints.brighton.ac.uk/16237/1/what%20community%20stories.pdf</u> [Accessed 10 July 2017].

WU, K.S. AND JOHNSTON, C.A., 2008. Hydrologic comparison between a forested and a wetland/lake dominated watershed using SWAT. *Hydrological Processes*, 22 (10), 1431-1442.

XAVIER, P. 2009. *Floodplain Woodland Hydrodynamics*. PhD Thesis, Hydroenvironmental Research Centre, Cardiff University. Cardiff, UK.

YARNELL, S. M., MOUNT, J. F. and LARSEN, E. W., 2006. The influence of relative sediment supply on riverine habitat heterogeneity. *Geomorphology*, 80, 310-324.

YORKSHIRE PEAT PARTNERSHIP, 2014a. *Grip blocking using peat dams.* Technical Specification 1. York: Yorkshire Peat Partnership.

YORKSHIRE PEAT PARTNERSHIP, 2014b. *Grip and small gully blocking using heather bale dams.* Technical Specification 2. York: Yorkshire Peat Partnership.

YORKSHIRE PEAT PARTNERSHIP, 2014c. *Larger grip and gully blocking using stone dams.* Technical Specification 3. York: Yorkshire Peat Partnership.

YORKSHIRE PEAT PARTNERSHIP, undated. *Gullies & large grips* [online]. York: Yorkshire Peat Partnership. Available from:

http://www.yppartnership.org.uk/restoration/gullies-and-large-grips/ [Accessed 19 July 2017].

ZELL, C., KELLNER, E. AND HUBBART, J.A., 2015. Forested and agricultural land use impacts on subsurface floodplain storage capacity using coupled vadose zone-saturated zone modeling. *Environmental Earth Sciences*, 74 (10), 7215-7228.

ZHANG, G.H., SHEN, R.C., LUO, R.T., CAO, Y. AND ZHANG, X.C., 2010. Effects of sediment load on hydraulics of overland flow on steep slopes. *Earth Surface Processes and Landforms*, 35 (15), 1811-1819.

ZHANG, L., DAWES, W.R., AND WALKER, G.R., 2001. Response of mean annual evapotranspiration to vegetation changes at catchment scale. *Water Resource Research*, 37 (3), 701-708.

ZHANG, M., WEI, X., SUN, P. AND LIU, S., 2012. The effect of forest harvesting and climatic variability on runoff in a large watershed: the case study in the Upper Minjiang River of Yangtze River basin. *Journal of Hydrology*, 464-465, 1-11.

ZHANG, Y., COLLINS, A.L. AND HODGKINSON, R., 2016. Use of farm survey returns from the Demonstration Test Catchments to update modelled predictions of sediment and total phosphorus loadings from subsurface drains across England and Wales. *Soil Use and Management*, 32 (S1), 127-137.

ZAO, Y., 2009. *Livestock impacts on hydrological connectivity*. PhD thesis, University of Leeds.

ZHENG, Z.C., HE, S.Q. AND WU, F.Q., 2012. Relationship between soil surface roughness and hydraulic roughness coefficient on sloping farmland. *Water Science and Engineering*, 5 (2), 191-201.

ZHOU, Q., 2014. A review of sustainable urban drainage systems considering the climate change and urbanization impacts. *Water*, 6 (4), 976-992.

ZHOU, X., LIN, H.S. AND WHITE, E.A., 2008. Surface soil hydraulic properties in four soil series under different land uses and their temporal changes. *Catena*, 73 (2),180-188.

List of abbreviations

AEP	annual exceedance probability
BACI	Before–After Control–Impact
CD	Chart Datum
CoGAP	Code of Good Agricultural Practice
CSE	catchment systems engineering
DHSVM	Distributed Hydrology Soil Vegetation Model
DTC	Demonstration Test Catchment
EIA	environment impact assessment
FCERM	flood and coastal erosion risk management
FEH	Flood Estimation Handbook
FIRM	Farm Integrated Run-off Management
FWAG	Farming and Wildlife Advisory Group
GIS	geographical information system
ICM	integrated crop management
IDB	Internal Drainage Board
JNCC	Joint Nature Conservation Committee
LWD	large woody debris
MMO	Marine Management Organisation
MOPS	Mitigation Options for Phosphorus and Sediment
NFM	Natural Flood Management
NGO	non-governmental organisation
NWRM	Natural Water Retention Measures
OD	Ordnance Datum
OMReg	Online Marine Registry
PPP	public-private partnership
RAF	run-off attenuation feature
RDP	Rural Development Programme
RSuDS	rural sustainable drainage system
SAC	Special Area of Conservation
SMP	Shoreline Management Plan
SPA	Special Protection Area
SRC	short coppice rotation

- SSSI Sites of Special Scientific Interest
- SuDS sustainable drainage system
- UKFS UK Forestry Standards
- VSA variable source area
- WWNP Working with Natural Processes

Would you like to find out more about us or about your environment?

Then call us on 03708 506 506 (Monday to Friday, 8am to 6pm)

email enquiries@environment-agency.gov.uk

or visit our website www.gov.uk/environment-agency

incident hotline 0800 807060 (24 hours) floodline 0345 988 1188 / 0845 988 1188 (24 hours)

Find out about call charges (www.gov.uk/call-charges)



Environment first: Are you viewing this on screen? Please consider the environment and only print if absolutely recessary. If you are reading a paper copy, please don't forget to reuse and recycle if possible.