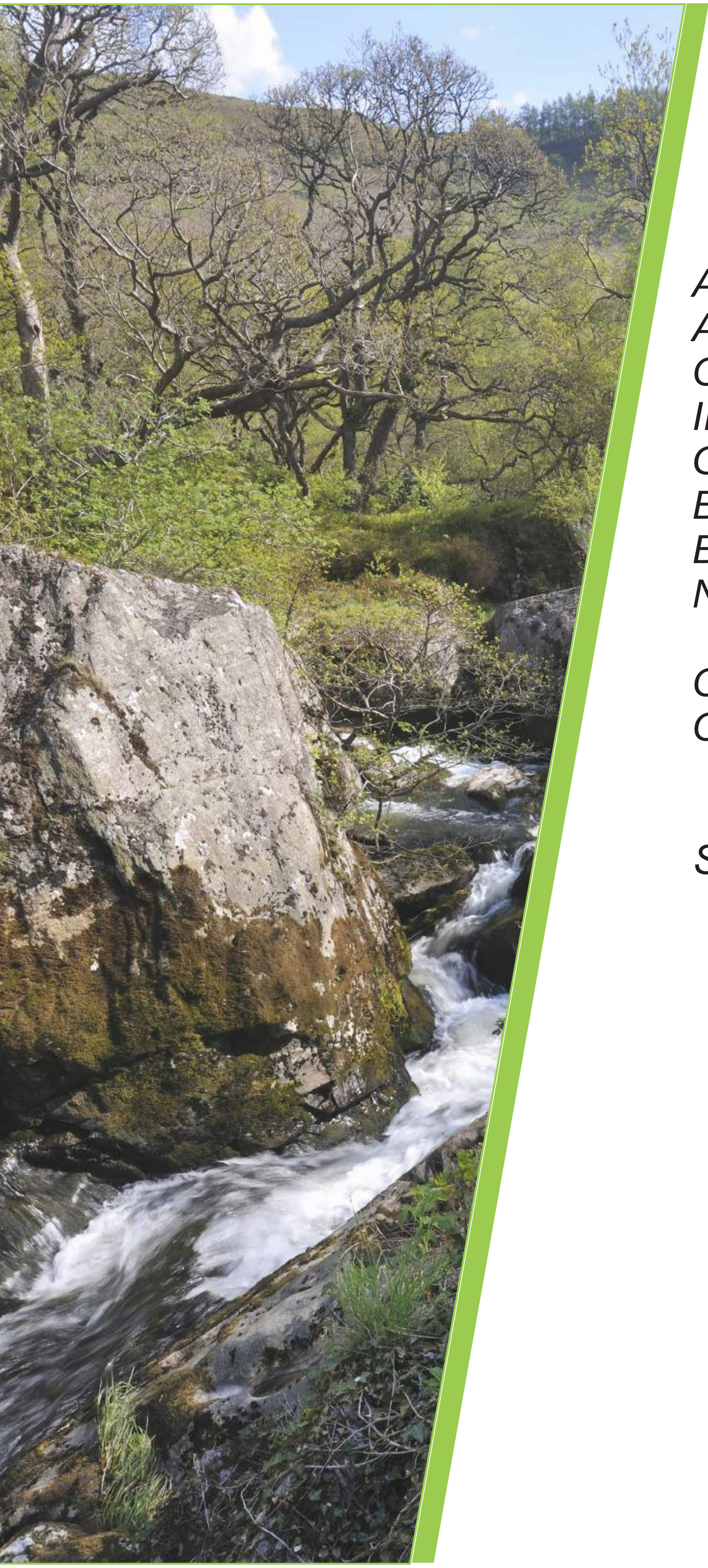

AECOM

AGGREGATE ASSESSMENT OF CLIMATE CHANGE IMPACTS ON THE GOODS AND BENEFITS PROVIDED BY THE UK'S NATURAL ASSETS

COMMITTEE ON CLIMATE CHANGE

SEPTEMBER 2015

UNIVERSITY OF EXETER

UNIVERSITY of York

Rev No	Comments	Prepared by	Checked by	Approved by	Date
3	Final Report	Andrew Suggitt University of Exeter Ilya Maclean University of Exeter Rob Critchlow University of York Colin Beale University of York Petrina Rowcroft AECOM Chris White AECOM	Steve Smith AECOM	Steve Smith AECOM	15 September 2015
2	Draft Final Report				28 August 2015
1	Draft Report (for Steering Group comment and internal and external peer review)				18 May 2015

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Job No 47072239 Date created: 09/03/2015

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Acknowledgements

The authors are very grateful to the numerous individuals and organisations that contributed to this research. In particular, we would like to extend our thanks to Tom August (CEH), Kathy Banner (NRW), Rob Christie (SEPA), Joanna Clark (University of Reading), Nathan Critchlow-Watton (SEPA), Kathleen Edson (EA), David Edwards (NIEA), Richard Fox (Butterfly Conservation), Simon Gillings (BTO), Dominic Habron (SEPA), Roy Haines-Young (University of Nottingham), Eleanor Hall (HR Wallingford), Amii Harwood (University of East Anglia), Joe Holden (University of Leeds), Carol Hrynkiewicz (Defra), Chris Huntingford (CEH), Brian Irvine (University of Leeds), Tom James (University of Exeter), Caroline Keay (NSRI), Gerald Leach (Yorkshire Water), Owen Lewis (EA), Allan Lilly (JHI), Colin Morice (UK Met Office), Christel Prudhomme (CEH), Matt Shepherd (NE), Stephen Sitch (University of Exeter), Steve Smith (University of Exeter), Pete Smith (University of Aberdeen), Kevin Walker (BSBI), UN FAO, UN CDAIC, UK Climate Impacts Programme, UK Met Office, and the Ordnance Survey.

We would also like to thank the following for helping to review the project, namely Mark Broadmeadow (FC), Iain Brown (James Hutton Institute), Mary Christie (SNH), Rebecca Clark (NE), Patrick Cowdy (Welsh Government), Amanda Crossfield (Yorkshire Water), Paul Dunhill (Welsh Government), Paul Fagan (Scottish Government), Julian Harlow (Defra), Stephen House (Defra), Nia Jackson-Jones (Welsh Government), Dewi Jones (Welsh Government), Jo-Anne Lawson (Scottish Government), Stephen McCabe (NIEL), Robert McCall (NRW), Michael McCallion (Department of the Environment Northern Ireland), Grainne Millar (Department of the Environment Northern Ireland), Michael Morecroft (NE), Harrier Orr (Environment Agency), Tasha Niesen (Defra), Zebedee Nii-Naate (Defra), Emma Phillimore (Defra), Helen Pontier (Defra), Judith Stuart (Defra), Sarah Taylor (Natural England), Clive Walmsley (NRW), Oliver Watts (RSPB), Sarah Webster (Defra), Rosalind West (Defra), Julian Wright (Environment Agency), and Graham Wynne (CCC, ASC).

We would also like to thank Dave Thompson, Kathryn Humphrey and Manuela DiMauro from the Committee on Climate Change for their invaluable help and assistance in guiding the project.

Abbreviations

AFE	Atlas Florae Europapeae
ASC	Adaptation Sub-Committee (to the Committee on Climate Change)
AWMN	Acid Waters Monitoring Network
BAP	Biodiversity Action Plan
BMAA	β-Methylamino-L-alanine
BOD	Biochemical Oxygen Demand
BRC	Biological Records Centre
BTO	British Trust for Ornithology
CABI	Commonwealth Agricultural Bureau International
CAPEX	Capital expenditures
CCC	Committee on Climate Change
CCIRG	Climate Change Impacts Review Group
CCRA	Climate Change Risk Assessment
CDAIC	Carbon Dioxide Information Analysis Center
CEH	Centre for Ecology and Hydrology
CEH LCM	CEH Land Cover Map
CIWEM	Chartered Institution of Water and Environmental Management
CO_2	Carbon dioxide
CRU	Climatic Research Unit (University of East Anglia)
CSF	Catchment Sensitive Farming
CSO	Combined Sewer Outflows
DECC	Department of Environment and Climate Change
DGVM	Dynamic Global Vegetation Model
DIC	Dissolved Inorganic Carbon
DO	Dissolved Oxygen
DOC	Dissolved Organic Carbon
DOENI	Department of the Environment Northern Ireland
EA	Environment Agency
EAA	European Economic Area
EC	European Commission
ECN	Environmental Change Network
ECOSSE	Estimating Carbon in Organic Soils Sequestration and Emissions (Scottish Executive)
EMBER	Effects of Moorland Burning on the Ecohydrology of River basins
EQS	Environmental Quality Standards
ETRS	European Terrestrial Reference System
EU	European Union
EU ETS	EU Emissions Trading System

EUPN	English Upland Peatland Network
EVRI	Environmental Valuation Reference Inventory
FC	Forestry Commission
FERA	Fera Science Limited, formerly the Food and Environment Research Agency
FFGWL	Future Flows and Groundwater Levels (CEH project)
FMA	Fertilisers Manufacturers' Association
GBP	Great British Pounds
GDDs	Growing Degree Days
GES	Good Ecological Status
GHG	Greenhouse Gas
GIS	Geographic Information System
HAB	Harmful Algal Bloom
IDW	Inverse Distance Weighting
IEEP	Institute for European Environmental Policy
INNS	Invasive Non-Native Species
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for Conservation of Nature
JHI	The James Hutton Institute
JRC	Joint Research Centre (EC)
Kt	Kilotonne
LAEA	Lambert Azimuthal Equal Area
LCMGB	Land Cover Map 1990 (CEH project)
LEAC	EAA Land Ecosystem Accounts
LULUCF	Land Use, Land Use Change and Forestry
MABSWE	Marlborough and Berkshire Downs and South-West Chilterns
MAC	Marginal Abatement Cost
MACC	Marginal Abatement Cost of Carbon
MIEX	Magnetic Ion-Exchange process
ML	Megalitre
MSFD	Marine Strategy Framework Directive
Mt	Megatonne
MTCO	Mean temperature of the coldest month
N	Nitrogen
NCC	Natural Capital Committee
NCEP	National Center for Environmental Prediction
NE	Natural England
NEA	National Ecosystem Assessment
NFU	National Farmers' Union
NGO	Non-Governmental Organisation
NIEA	Northern Ireland Environment Agency

NIEL	Northern Ireland Environment Link
NIR	National Inventory Report
NIWT	National Inventory of Woodlands and Trees
NNR	National Nature Reserve
NNSS	Non-Native Species Strategy
NPV	Net Present Value
NRW	Natural Resources Wales
NSI	National Soils Inventory
NSRI	National Soil Resources Institute
NTPC	Non-Traded Price of Carbon
NTSLF	National Tidal and Sea Level Facility
NVZ	Nitrate Vulnerable Zone
OECD	Organisation for Economic Co-operation and Development
ONS	Office for National Statistics
OSGB	Ordnance Survey Great Britain
P	Phosphorus
PAHs	Polycyclic Aromatic Hydrocarbons
PES	Payment for Ecosystem Services
PgC	Petagram of Carbon
RBD	River Basin District
RBMP	River Basin Management Plan
RCM	Regional Climate Model
RCP	Representative Concentration Pathway
RSPB	Royal Society for the Protection of Birds
SCaMP	Sustainable Catchment Management Programme
SCC	Social cost of carbon
SCP	Spatially Coherent Projections (UKCP product)
SEA	Strategic Environmental Assessment
SEPA	Scottish Environment Protection Agency
SNH	Scottish Natural Heritage
SNIFFER	Scotland and Northern Ireland Forum for Environmental Research
SRES	Special Report on Emissions Scenarios (IPCC product)
SRUC	Scotland’s Rural College
SSSI	Site of Special Scientific Interest
SuDS	Sustainable urban Drainage Systems
SWMI	Significant Water Management Issue
tC	Tonnes of carbon
THMs	Trihalomethanes
TOC	Total Organic Carbon
TPC	Traded Price of Carbon

TTSS	Thames Tideway Strategic Study
UKCIP	UK Climate Impacts Programme
UKCP09	UK Climate Projections 2009
UKNEA-FO	UKNEA Follow-On
UKTAG	UK Technology for Agriculture and Genetics
UN	United Nations
UNCED	United Nations Conference on Environment and Development
UNEP	United Nations Environment Programme
UNFAO	United Nations Food and Agriculture Organisation
UNFCCC	United Nations Framework Convention on Climate Change
UWMN	Upland Waters Monitoring Network
WAG	Water Appraisal Guidance
WEI	Water Exploitation Index
WFD	Water Framework Directive
WHO	World Health Organisation
WTA	Willingness to accept
WTP	Willingness to pay
WWTD	Urban Wastewater Treatment Directive

Executive summary

Introduction

The Government is required to publish a Climate Change Risk Assessment (CCRA) every five years under the Climate Change Act 2008. The purpose of the CCRA is to provide an assessment of the current and future risks and opportunities to the UK from climate change and inform priorities for the next cycle of national adaptation programmes for England, Northern Ireland, Scotland and Wales. The Adaptation Sub-Committee (ASC) to the Committee on Climate Change (CCC) has a statutory duty to advise Ministers on preparation of the CCRA. The first CCRA was published in 2012 and the next is due in 2017. This second CCRA ('CCRA2') will take account of barriers to, and opportunities to facilitate, adaptation. For CCRA2, Ministers have asked the ASC to provide their statutory advice in the form of an independent Evidence Report which will inform a Government Report to be laid before Parliament. The Evidence Report must be submitted to Ministers by July 2016. In response to the risks identified by CCRA2, the Government and the Devolved Administrations will develop second National Adaptation Programmes.

AECOM – in partnership with the University of Exeter, the University of York, Scotland's Rural College (Dominic Moran) and Ricardo-AEA (Richard Smithers) – was commissioned to undertake research to support the preparation of the CCRA Evidence Report. Four research projects were commissioned to support the Evidence Report and the AECOM-led consortium delivered Project C:

- Project A – Updated projections of flood risk for the UK
- Project B – Updated projections of water availability for the UK
- **Project C – Aggregate assessment of climate change impacts on the goods and benefits provided by the UK's natural assets**
- Project D – Development of high-end scenarios for a number of climate impacts beyond sea level rise/storm surge

The research focused on a series of goods derived from natural assets which the Natural Capital Committee (NCC), in its second report, considered to be at very high risk:

- clean water from mountains, moors and heaths, due to the deteriorating quality of those habitats (e.g. peatland degradation);
- clean water from the current extent and projected growth of urban areas leading to a deterioration in freshwater, soils and natural water purification processes in these areas;
- wildlife in several land use categories (semi-natural grasslands, enclosed farmland and freshwaters) due to poor quality habitats and unfavourable 'spatial configurations'; and
- equable climate as a result of the degraded condition of mountains, moors and heaths which have the potential for much greater carbon storage.

Climate change could exacerbate pressures on natural assets and potentially the goods and benefits they ultimately provide. The aim of Project C was to assess the impact of climate change under different climate projections on clean water, equable climate and wildlife and endeavour to do so based on a quantitative, spatially explicit methodology. In practice, it was only possible to do so for equable climate and wildlife. With respect to clean water, in discussion with the CCC, it was decided that a literature review would provide a better means to inform the ASC's independent Evidence Report. In addition to the climate change impact assessment, an effort was made to place a monetary value on the costs associated with changes in clean water, carbon storage and wildlife and associated benefits that could be attributed to climate change; however, this proved very challenging in practice and was only partially possible with respect to equable climate.

Clean water

Having established the present day condition of the water environment drawing primarily on Water Framework Directive data, the literature review for clean water considered climate change impacts with respect to five existing risks to the provision of clean water:

- Nutrient enrichment and eutrophication
- Combined sewer overflows
- Dissolved Organic Carbon
- Specific pollutants, priority substances, and 'other' chemical pollutants
- Over-abstraction and saline intrusion

For each of the five risks, the report provides:

- a description of the problem in which the causes and mechanisms are discussed;
- the trend in the risk, assessed as increasing (↑), decreasing (↓), or stable (–),followed by a qualitative judgment of confidence in these trends ('high', 'medium', 'low');
- the impacts of climate change on the risk;
- adaptation measures to reduce the risk;
- a case study of the monetary value of the costs associated with changes in clean water provision (NB except for 'specific pollutants, priority substances, and 'other' chemical pollutants)

Climate change exerts less of an influence over nutrient enrichment and eutrophication than other factors. Nevertheless, climate change could exacerbate existing (natural and anthropogenic) drivers of nutrient enrichment with changes in the seasonality of precipitation a critical factor. For example, there will be less dilution of nutrients in summer if projected changes to precipitation are accurate with reduced scope for dilution at point sources a particular concern. Table 0-1 provides an overview of climate change impacts on nutrient enrichment and eutrophication.

Table 0-1 Overview of climate change impacts on nutrient enrichment and eutrophication

Direction of impact	Climate variable	Present trend (confidence)	Future trend (confidence)	Mechanism(s)	Relative Importance
Exacerbating	Temperature	↑ (High)	↑ (High)	Accelerated growth of algae; lower dissolved oxygen	+
	Winter precipitation	↑ (Low)	↑ (Low)	Soil erosion; nutrient cycling	++
	Summer precipitation	↓ (Low)	↓ (Medium)	Less dilution; longer residence times	++
	Drought	– (Low)	↑ (Low)	Increased mineralisation	+
	Storminess	- (High)	↑ (Medium)	Increased Combined Sewer Overflows (CSOs); increased nutrient load from agriculture	+
Mitigating	Annual precipitation	– (Low)	↓ (Low)	Reduced runoff, longer residence times	+

With respect to combined sewer overflows (CSOs), there is some evidence that both the duration and frequency of CSO discharge events will increase, but that the sensitivity of these systems to climate change depends on their ability to store excess water. Table 0-2 provides an overview of climate change impacts on CSOs.

Table 0-2 Overview of climate change impacts on CSOs

Direction of impact	Climate variable		Mechanism(s)	Relative Importance
	Present trend (confidence)	Future trend (confidence)		
Exacerbating	Winter precipitation		Increased discharge and runoff to sewage systems	++
	↑ (Low)	↑ (Low)		
	Summer precipitation		Reduced flushing and dilution	+
	↓ (Low)	↓ (Medium)		
	Rainfall intensity		Flashier, more frequent runoff	++
	↑ (Low)	↑ (Medium)		
	Temperature		Reduced assimilative capacity of environment	+
	↑ (High)	↑ (High)		

The decomposition of peatland habitats leads to the release of organic carbon from soils. Dissolved Organic Carbon (DOC) can affect water colouration, the toxicity of certain metals and result in changes in ecological conditions. The available evidence suggests that DOC concentrations have risen and, while various drivers of DOC release have been proposed, the increase is most likely attributable to decreased acid deposition (‘acid rain’, but also dry deposition). With acid deposition appearing to stabilise, the main determinants of DOC levels in future may increasingly become climate and land management. Table 0-3 provides an overview of climate change impacts on DOC levels.

Table 0-3 Overview of climate change impacts on DOC levels

Direction of impact	Climate variable		Mechanism(s)	Relative Importance
	Present trend (confidence)	Future trend (confidence)		
Exacerbating	Drought		Water table draw down, leaving soil carbon more vulnerable to being mobilised	++
	↑ (Low)	↑ (Low)		
	Storminess		Increased dissolution (and suspension) of soil carbon and transport into streams and rivers	++
	- (High)	↑ (Medium)		
	Temperature		Increased soil production of DOC; reduced frost days and snow would increase exposure to ultraviolet photodegradation, leading to higher runoff; increased frequency of fires would also leave soil carbon vulnerable to transport via runoff	+
	↑ (High)	↑ (High)		

Mitigating	Winter precipitation		More winter precipitation (if not intense) would increase wetness and decrease vulnerability of soil carbon to mobilisation	+
	↑ (Low)	↑ (Low)		

With respect to specific pollutants, priority substances, and 'other' chemical pollutants, most WFD failures for these pollutants occur in urban or ex-mining contexts, with multiple, diverse origins. The impact of climate change on the risk from these pollutants will be dependent on changes in precipitation at particular times of year, although rising temperatures could mitigate the risk by increasing degradation rates; however, the net outcome of the volatilisation may not necessarily be positive, as the newly gaseous chemicals could harm wildlife and crops. Table 0-4 provides an overview of climate change impacts on specific pollutants, priority substances, and 'other' chemical pollutants.

Table 0-4 Overview of climate change impacts on specific pollutants, priority substances, and 'other' chemical pollutants

Direction of impact	Climate variable		Mechanism(s)	Relative Importance
	Present trend (confidence)	Future trend (confidence)		
Exacerbating	Summer precipitation		Concentration	++
	↓ (Low)	↓ (Medium)		
	Storminess		Direct transport; Indirect via CSOs	+
	- (High)	↑ (Medium)		
Mitigating	Winter precipitation		Increased dilution	++
	↑ (Low)	↑ (Low)		
	Temperature		Volatilisation; degradation	+
	↑ (High)	↑ (High)		

The adverse impacts of abstraction can encompass all the negative impacts associated with low flows, such as higher river temperatures, fish stress and mortality, reductions in invertebrate density, and increased concentrations of other pollutants (NB 'over-abstraction' refers to abstraction that results in harm to the water environment, rather than that which is considered to be over-licenced under Defra's licensing regime). Saline intrusion can result from over-abstractions from groundwater supply (via pumps, boreholes or wells), wherein the hydraulic gradient from the land to the sea is weakened, and sometimes reversed, by the removal of freshwater. Table 0-5 provides an overview of climate change impacts on over-abstraction and saline intrusion.

Table 0-5 Overview of climate change impacts on over-abstraction and saline intrusion

Direction of impact	Climate variable		Mechanism(s)	Relative Importance
	Present trend (confidence)	Future trend (confidence)		
Exacerbating	Sea level rise		Thermal expansion of the oceans will continue to drive sea level rise and increase salt load	+
	↑ (High)	↑ (High)		
	Summer precipitation		Decreases in summer precipitation will reduce groundwater recharge and increase rates of abstraction	++
	↓ (Low)	↓ (Medium)		

	Temperature		Higher temperatures will increase demand for water, particularly during the summer months	++
	↑ (High)	↑ (High)		
	Drought		More droughts would increase abstraction as other (surface) sources of water dry up	++
	↑ (Low)	↑ (Low)		
Mitigating	Winter precipitation		More winter precipitation (if not intense) would increase groundwater recharge and decrease the risk of over-abstraction	+
	↑ (Low)	↑ (Low)		

Equable climate

With respect to equable climate, we have demonstrated how three different socio-economic directions result in substantially different outcomes for the soil and vegetation carbon stock of Great Britain. In the case of soil carbon stock, potential gains ranged from +6% to +11% (across two scenarios: 'Local stewardship', and 'Green and pleasant land'), while potential losses ranged from –12% to –13% (under 'World markets'). The corresponding monetary value of these changes in soil carbon also ranged widely: from a net present value of £39-74 billion across the 'Local stewardship' and 'Green and pleasant land' scenarios, to a loss of £83-88 billion under 'World markets'.

In the case of vegetation carbon stock, scenario uncertainty was even higher, ranging from +23% to +28% gains ('Green and pleasant land') to –3% to –8% losses ('World Markets'). While the monetary values ranged from a net present value of £8-10 billion ('Green and pleasant land') to a loss of £1-3 billion ('World Markets'). The indirect effects of climate change were more pronounced on vegetation carbon stocks than on soil carbon stocks, heavily modifying the estimated net changes in vegetation carbon under 'Local stewardship' and 'World Markets'. In the latter scenario, the indirect effect on arable land was on a par with the effect of land cover change itself.

These headline results mask further spatial variation in the changes to stock we estimated, with land cover changes and climate change affecting the constituent countries of the UK to varying degrees. A comparison of the spatial variation in the value of soil and vegetation carbon is set out in Table 0-6. Because England in particular is subject to a substantial number of drivers acting simultaneously (urbanisation, climate change, agricultural changes), uncertainties generated by differing socio-economic and climate futures resulted in larger variations in estimated carbon amounts. Wales and Scotland are subject to less pressure from these drivers, and thus estimates for these countries were less variable. These findings highlight the importance of applying a spatial approach in national assessments of natural assets. Note that the analysis did not encompass Northern Ireland due to data constraints.

Table 0-6 Comparison of the spatial variation in the value of soil and vegetation carbon stocks from 2010 to 2060

Region	Local stewardship (low emissions)	Local stewardship (high emissions)	Green & pleasant land (low emissions)	Green & pleasant land (high emissions)	World markets (low emissions)	World markets (high emissions)
Great Britain						
Soil	£39 billion	£43 billion	£58 billion	£74 billion	-£88 billion	-£83 billion
Vegetation	-£87 million	£2 billion	£10 billion	£8 billion	-£3 billion	-£1 billion
England						
Soil	£22 billion	£24 billion	£38 billion	£50 billion	-£33 billion	-£29 billion
Vegetation	£253 million	£2 billion	£8 billion	£7 billion	-£2 billion	-£49 million
Scotland						
Soil	£10 billion	£11 billion	£12 billion	£13 billion	-£47 billion	-£46 billion
Vegetation	-£328 million	-£204 million	£663 million	£602 million	-£709 million	-£630 million
Wales						
Soil	£4 billion	£5 billion	£4 billion	£5 billion	-£8 billion	-£8 billion
Vegetation	-£3 million	£17 million	£200 million	£152 million	-£120 million	-£104 million

It must be noted that all our estimates represent equilibrium changes, and would in reality take anything from 50 to 750 years to occur (UK NIR 2014). This has important implications for the provision of an equable climate this century. In the case of vegetation carbon, the means of managing forests for carbon are becoming clearer (Forest Research 2012), and thus woodlands could play an important role in the UK's provision of an equable climate. The evidence for the efficacy of techniques to sequester carbon in the soil is more equivocal (Thomson et al. 2012), but because the potential gains are substantial, research will continue to focus on how these gains might be achieved. As ever, it will be necessary to balance any ambition to increase carbon stock with existing uses of the land (e.g. farming), where priorities can sometimes differ. Attention will therefore focus on approaches that bring multiple benefits (e.g. restoring upland peats for carbon, water and wildlife).

Wildlife

The wildlife assessment considered climate change impacts on more than 4000 species from 17 taxonomic groups, providing a broad spatial assessment of the potential impacts on biodiversity across Great Britain and Northern Ireland. The assessment drew on state-of-the-art species distribution models that account for recording effort and can quantify impacts of unknown variables, e.g. land-use change.

The results indicate high species-specific sensitivity to climate change, but systematic differences between taxonomic groups. Bryophytes (non-vascular plants which include mosses, liverworts and hornworts) are the group most likely to be negatively impacted by climate change, and are also of high international conservation importance. Along with bryophytes, a high proportion of ants are noticeably more sensitive to a changing climate than other taxa, but are more likely to have increase in spatial distribution. There is relatively little difference in the proportions of taxa in different risk categories associated with 2, 4 or 6˚C global warming scenarios projected to 2070-2099 in comparison to changes against the 1960-90 baseline: most of the anticipated changes happen in the UK with relatively little climate change, although there may be species unaffected by low levels of climate change that are impacted by changes at 4 or 6˚C change.

The greatest spatial change in species distributions amongst all taxonomic groups occur in northern and upland areas. However, this analysis does not take into account species colonising from the south of the UK, the impact of environmental refugia potentially providing protection from broader-scale climate changes, and other indirect impacts of climate change such as changing land-use policies. Additional analyses focussing only on the impact of climate variables on potential change in climate space revealed similar spatial patterns of change to the full analysis.

Protected areas are likely to see relatively more species turnover, i.e. more local losses and local colonisations, in comparison to surrounding land. This is likely to reflect the location of protected areas in areas of unusual or marginal environments, it are these environments where climate change has a greater impact on species presence in comparison to areas where generalist species are more abundant.

Priority areas for conservation have a different distribution under current climatic conditions than in the future, Additional protected areas, particularly in northern and western parts of the UK, could potentially improve protection in the future.

With respect to valuation, given the difficulties in deriving robust values for the non-use values of wildlife and the fact that focusing on economic species (or some sub-set thereof) would only give a partial picture of the change, no attempt was made to value the impacts of climate change on wildlife in light of the wildlife modelling. However, a commentary was provided on what such a valuation would require in terms of data.

Overall conclusions

Overall, the report provides a comprehensive review of climate risks to water quality, which emphasises that climate change will be an increasingly important challenge to meeting environmental targets. With regards to equable climate, the report presents an original and innovative assessment of the risks and opportunities to UK carbon stores from climate-induced changes in land cover, which highlights that UK's the carbon stores could be significantly degraded under a high-growth scenario. With respect to wildlife, the report provides improved and updated modelling of changes we can expect in the climate space of native British wildlife, which highlights that mosses (bryophytes) are particularly at risk of being unable to respond to changing climatic conditions, which could have implications for the key ecosystem services they provide (i.e. water purification, carbon storage, flood alleviation) and global consequences as the UK hosts internationally important populations of these taxa.

Table 0-7 below summarises the overall risks to clean water, equable climate and wildlife identified through the research for the constituent parts of the UK.

Table 0-7 Risks to clean water, equable climate and wildlife

Clean water	Equable climate	Wildlife
England		
Over-abstraction a particular risk for South-east England, although groundwater recharge could be enhanced by increased winter precipitation. Continued DOC export likely to continue as upland areas recover from acidification. Future increases in rainfall intensity would increase the risk of CSO discharges, particularly in urban areas without SUDs installed. Nutrient enrichment could interact with temperature-related productivity increases to cause more algal blooms.	Of all the UK constituent countries, carbon stocks in England have the highest exposure to the changes in land use set out in NEA (2011). This is due to the effect of urbanisation and the abandonment of arable land, particularly in the South and East of England. High uncertainty arising from socio-economic change.	A contrast between northern and southern areas of the country, with northern areas expected to see greater colonisations and extinctions. However, the analysis does not account for colonists to the south. A high number of additional SPAs could improve protections for bird species in the future, particularly on the coast of East Anglia and the south west.
Northern Ireland		
Projected changes in precipitation (both seasonal and annual) are lower for NI than for the other constituent countries of the UK. Thus the potential for these changes to exacerbate existing risks to clean water is lower. Existing risk from nutrient enrichment will be enhanced by projected temperature increases.	*No data.*	In comparison to the other countries there is likely to be the lowest numbers of colonization and extinctions. For all three protected areas networks, a high number of additional protected areas, particularly on in coastal areas, are likely to improve species protection for UK species.
Scotland		
Scotland less exposed to projected declines in summer precipitation, thus less susceptible than other UK constituent countries to low flows, over-abstraction and exacerbation of nutrient enrichment. Lower projected increases in temperature mean that Scotland also faces a lower risk of increased soil production driving DOC export.	The potential for equilibrium losses or gains in carbon stock is lowest of the UK constituent countries. Reversions to semi-natural grassland (from enclosed farmland) provide opportunities for equilibrium increases in soil carbon. Abandonment of unproductive agricultural land provides a benefit to vegetation carbon stocks in upland areas, via natural succession. Lower uncertainty arising from socio-economic change.	Mainland areas are likely to see the highest numbers of colonisations and extinctions (i.e. greatest turnover) in comparison to the rest of the UK, particularly on the west coast. In contrast, the western and northern isles have some of the lowest expected species turnover. Zonation analysis for the whole of the UK suggests that the majority of new protected areas in the future should be located in Scotland.
Wales		
High contrast between projected increases in winter precipitation and decreases in summer precipitation in Wales. Both these changes will exacerbate nutrient enrichment, via increased soil erosion and lower dilution rates, respectively. They will also lead to an increased risk of CSO discharges. Risk of over-abstraction of water in East Wales.	The potential for equilibrium loss of soil carbon is relatively high for UK constituent countries, with intensification of farming posing a particular risk for semi-natural and wild habitat in the Welsh Mountains. Vegetation carbon at risk in these upland areas across most socio-economic scenarios. Lower uncertainty arising from socio-economic change.	Upland areas are likely to see the greatest levels of colonisations and extinctions, with coastal areas having a relatively lower species turnover. Prioritising areas for new and expansion of protected area networks across the UK resulted in very few additional priority areas located within Wales.

In light of the research, some specific adaptation measures are highlighted; for example,

- A catchment-based approach to improving water quality, i.e. the scale at which multiple land owners/managers as well as statutory and third sector actors can be engaged with a common goal in mind (e.g. reducing diffuse pollution).

- Improved on –farm nutrient management through, for example, changes in agricultural practice (e.g. avoiding the application of fertiliser when the weather or soil conditions are likely to lead to runoff); improved compliance (e.g. with respect to septic or slurry tank storage); the creation of riparian buffer strips; and various measures to decrease soil erosion.

- Targeting point source pollution from sewage treatment works including through, for example, reducing or removing phosphate from detergent products through regulation and 'stripping' phosphate from sewage effluent as part of the treatment process.

- Reducing discharges from CSOs through, for example, creating separate sewage systems for rainwater and wastewater, implementing Sustainable Urban Drainage Systems (SUDS) which divert runoff from the sewage system, and promoting measures to reduce domestic water use.

- Addressing increasing DOC levels through restoring damaged or degraded peatlands, including through promoting innovative payments for ecosystem services (PES) schemes.

- Targeted measures to address specific pollutants, priority substances and other chemicals such as ultraviolet (UV) disinfection at sewage treatment works (many of the measures outlined above will also help target specific pollutants, priority substances and other chemicals)

- Limiting the effects of over-abstraction through changes to the abstraction licensing regime and reducing abstraction pressure through investment in on-farm reservoir capacity and improved soil management.

- Improved soil management to optimise carbon sequestration potential (although the evidence for the efficacy of techniques such as reducing or stopping tillage is equivocal)

- Increasing the coverage of SSSIs in the future from 10% to 17% of the land area could increase UK species protection by 5%.

However, more broadly, the research indicates a need to pursue adaptation measures that result in natural asset restoration or enhancement at the landscape scale and, in doing so, address numerous pressures simultaneously and yield multiple benefits. The Natural Capital Committee (2015) in its third report proposed a series of investments in natural capital, several of which would promote landscape scale interventions, help alleviate multiple pressures and generate numerous benefits, for example:

- woodland planting of up to 250,000 additional hectares;

- peatland restoration on around 140,000 hectares in upland areas; and

- wetland creation on around 100,000 hectares

Crucially, implementing natural capital restoration on this scale would help to alleviate many of the anticipated climate change pressures documented in this report. For example, these interventions could help to improve soil management which aids water regulation and can help to reduce soil carbon losses. Woodland planting, in particular, can help to regulate water flows and quality, reduce soil erosion and provide habitat for wildlife.

The NCC also recommends that government, working with business, NGOs and other parts of society, should develop a 25 year plan to protect and enhance natural capital; this plan provides an obvious vehicle to enact the landscape-scale adaptation measures discussed above and could include climate change adaptation as a primary aim.

Introduction

1 Introduction

Background

1.1.1 The Government is required to publish a Climate Change Risk Assessment (CCRA) every five years under the Climate Change Act 2008. The purpose of the CCRA is to provide an assessment of the current and future risks and opportunities to the UK from climate change and inform priorities for the next cycle of national adaptation programmes for England, Northern Ireland, Scotland and Wales. The Adaptation Sub-Committee (ASC) to the Committee on Climate Change (CCC) has a statutory duty to advise Ministers on preparation of the CCRA. The first CCRA was published in 2012 and the next is due in 2017. This second CCRA ('CCRA2') will take account of barriers to, and opportunities to facilitate, adaptation. For CCRA2, Ministers have asked the ASC to provide their statutory advice in the form of an independent Evidence Report which will inform a Government Report to be laid before Parliament. The Evidence Report must be submitted to Ministers by July 2016. In response to the risks identified by CCRA2, the Government and the Devolved Administrations will develop their own separate adaptations programmes.

This project

1.1.2 AECOM – in partnership with the University of Exeter, the University of York, Scotland's Rural College (SRUC) and Ricardo-AEA – was commissioned to undertake research to support the preparation of the CCRA Evidence Report. Four research projects were commissioned in support of the Evidence Report and the AECOM-led consortium delivered Project C:

- Project A – Updated projections of flood risk for the UK
- Project B – Updated projections of water availability for the UK
- **Project C – Aggregate assessment of climate change impacts on the goods and benefits provided by the UK's natural assets**
- Project D – Development of high-end scenarios for a number of climate impacts beyond sea level rise/storm surge

1.1.3 The Natural Capital Committee (NCC) was established to advise Government on the state of natural capital in England. The NCC's second report on the state of natural capital anticipated that pressures on natural capital, including from population growth and the consequent increase in demand for food, housing and transport, would persist and intensify (Natural Capital Committee 2014). These pressures can affect natural capital and undermine the benefits it provides us with.

1.1.4 Natural capital assets come together in a variety of ways to ultimately generate benefits (Natural Capital Committee 2014). Assets acting in combination provide ecosystem services which are, in turn, combined with other types of capital (financial, human, manufactured and social) to produce goods; these goods are then used or consumed and thus provide benefits to people. For example, freshwaters (an asset) provide a flow of clean water (a service), which can be treated to provide drinking water (a good) to support human wellbeing (a benefit) (Natural Capital Committee 2014). This conceptual framework is illustrated in Figure 1-1.

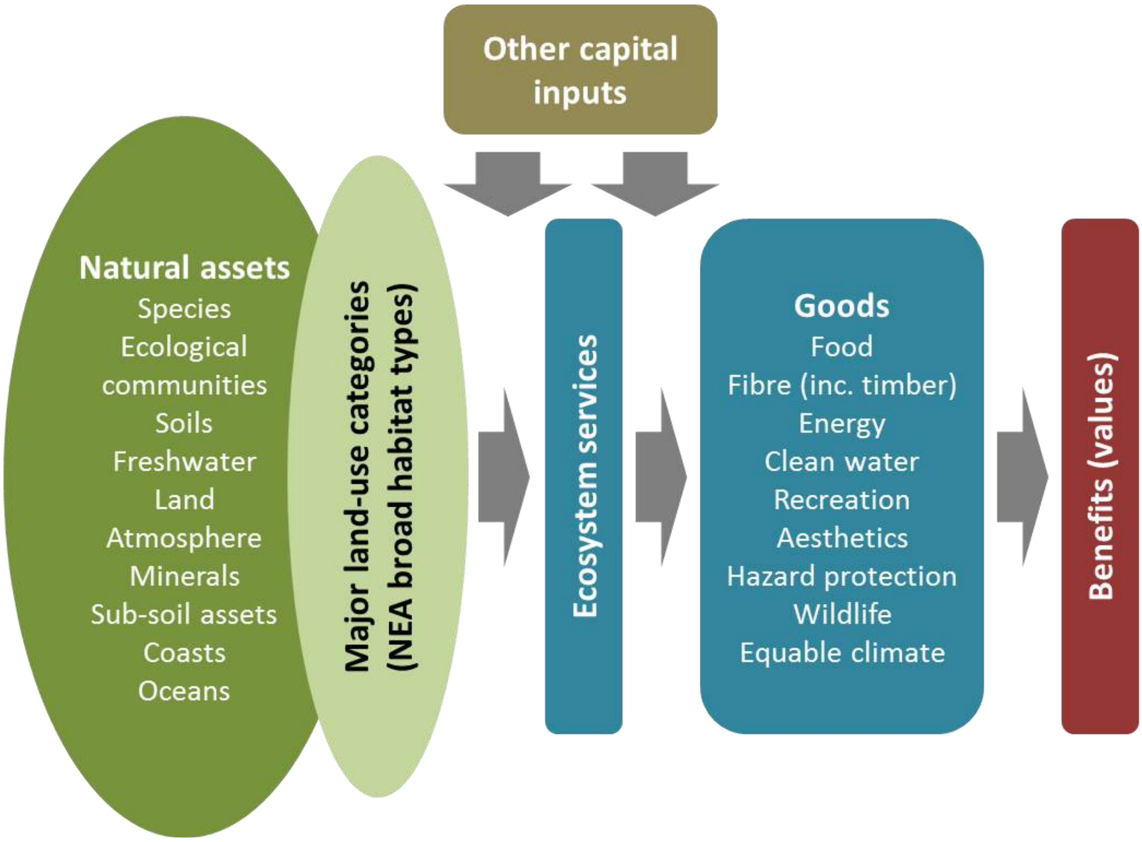


Figure 1-1 Natural capital and benefits to people – a framework

(Source: Natural Capital Committee, 2014.)

1.1.5 The NCC's second report identified a series of benefits considered to be at high or very high risk – see Figure 1-2. Those at **very high risk** were:

- **clean water** from mountains, moors and heaths, due to the deteriorating quality of those habitats (e.g. peatland degradation);
- **clean water** from the current extent and projected growth of urban areas leading to a deterioration in freshwater, soils and natural water purification processes in these areas;
- **wildlife** in several land use categories (semi-natural grasslands, enclosed farmland and freshwaters) due to poor quality habitats and unfavourable 'spatial configurations' ; and
- **equable climate** as a result of the degraded condition of mountains, moors and heaths which have the potential for much greater carbon storage.

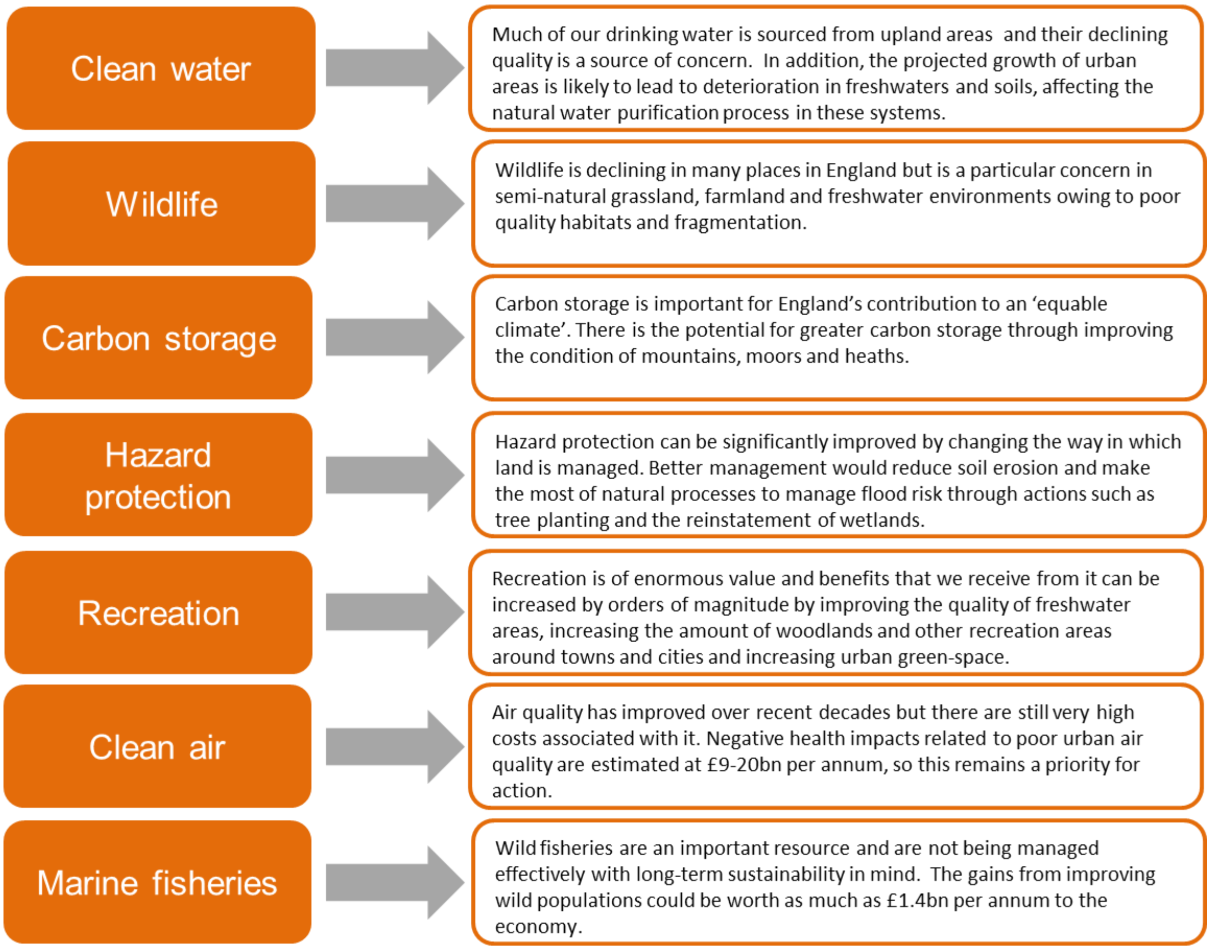


Figure 1-2 Benefits from natural capital at high or very high risk for England

(Source: Natural Capital Committee, 2014)

1.1.6 Climate change could exacerbate pressures on natural assets and potentially the benefits they ultimately provide. The aim of Project C was to assess the impact of climate change under different climate projections on the goods and benefits provided by the UK's natural assets. More specifically, the aim was to determine the extent to which climate change, in combination with non-climate pressures (e.g. population growth and land cover change), might impact on important goods which might, in turn, affect the quality and availability of certain benefits.

Specific focus

1.1.7 In light of the NCC's analysis, a decision was taken early on to focus on the goods identified by the NCC to be at very high risk, namely **clean water**, **equable climate** (essentially carbon storage) and **wildlife** (see above). Furthermore, in line with the CCC's aspiration that the research should take a spatial approach to mapping the goods and benefits provided by natural assets and the ways in which these might be impacted by climate change, the project team endeavoured to establish a quantitative, spatially explicit methodology for assessing climate change impacts on clean water, equable climate and wildlife. However, in practice, it was only possible to do so for equable climate and wildlife. With respect to clean water, in discussion with the CCC, it was decided that a literature review would provide a better means to inform the independent Evidence Report being prepared by the ASC. Furthermore, it proved very difficult to incorporate analysis of non-climate pressures (population growth, land cover change etc.) such that the impacts of climate change on the goods and benefits provided by natural assets could be confidently attributed to climate change and non-climate pressures, respectively. However, the assessment of climate change for equable climate did incorporate consideration of future land cover change.

1.1.8 It was originally envisaged that, having undertaken quantitative, spatially explicit analysis, it would be possible for each of the three areas of focus – clean water, equable climate and wildlife – to place a monetary value on the costs associated with changes in the availability and quality of goods and benefits that could be attributed to climate change. However, this proved very challenging in practice. With respect to clean water, with the switch away from a quantitative, spatially explicit approach, the focus of the valuation section was on presenting the evidence relating to the (monetary) value of changes in water quality. With regard to equable climate, it was possible to derive some broad estimates of the value of changes in soil and vegetation carbon stocks under different climate and land use scenarios. In relation to wildlife, in discussion with CCC and the project Steering Group, there was no attempt to undertake valuation and, instead, a commentary was provided highlighting the challenges.

1.1.9 We used case studies in the sections on water quality and wildlife to illustrate the significance (in monetary terms) of the potential impacts of climate change on the goods and benefits provided by these goods. However, it is important to note that the values presented in the cases studies may not necessarily be representative of the situation across the UK; that is because they are often derived in a specific context and represent the preferences of a particular population for a defined change in ecosystem provision from a defined baseline situation. Nevertheless, there are established techniques such as value transfer (also known as 'benefits transfer') that allow existing economic valuation evidence to be applied in a new context. Value transfer is typically a quicker and lower cost approach to generating economic valuation evidence, compared to commissioning a specifically designed primary valuation study and it therefore offers a practical tool for estimating values in cases where it is not feasible to undertake new field studies (eftec 2010). The value transfer literature embraces a number of approaches including mean value transfer, meta-analyses to inform regression models linking values to the characteristics captured in the available source data and the estimation of more sophisticated spatially explicit value functions (Bateman et al. 2011).

1.1.10 While there are a number of issues associated with the use of value transfer, these are well understood and largely accepted. However, care needs to be taken to ensure that any values used are adjusted where appropriate, judiciously applied and that any caveats and uncertainties are clearly highlighted in the ensuing analysis. Defra's Value Transfer Guidelines (eftec 2010) set out some of the key factors to be considered in valuing environmental impacts using value transfer. These include:

- the characteristics of the good likely to influence its economic value (e.g. size, location, scarcity, unique features);
- the size of the affected or beneficiary population;
- the economic and environmental baseline conditions;
- the nature, scale and direction of change; and
- the quality of the monetary valuation evidence.

1.1.11 Speed and cost make value transfer a practical tool for policy appraisal given the constraints on time and resources available for decision-making.

Climate trends and projections

1.1.12 In order to provide context for the assessment and valuation work to follow, the next section provides an overview of trends and projections for three climate variables: temperature and precipitation; extreme events; and sea level rise.

Temperature and precipitation

1.1.13 Every part of the UK has warmed since 1960, experiencing approximately a degree of warming in later decades (1°C since 1980 in England and Wales, 0.8°C in Scotland and Northern Ireland, 0.7°C in coastal areas, Jenkins et al. 2008). There is a NW/SE gradient in the spatial patterning of this warming, such that London and the South-East has experienced more warming than Scotland and Northern Ireland. This gradient exists at all times of year, although is slightly stronger in winter and summer (Figure 1-3 a). The warming trend is very likely to continue across the country, particularly during the summer, and for the South of England (Murphy et al. 2009).

1.1.14 There has been no consistent recent trend in annual precipitation falling on the UK (Hulme et al. 2002, Jenkins et al. 2008), although there are some indications that the seasonality of precipitation has shifted towards drier summers and wetter winters in some regions (Marsh et al. 2007, Jenkins et al. 2008, Watts et al. 2013, Figure 1-3 b). Winters are projected to get wetter, particularly in Scotland (under central estimates, medium emissions scenario, 2080s), although less so on the higher ground of the North Pennines and the Scottish Highlands (Murphy et al. 2009). A 10 - 40% decrease in summer precipitation is projected for most of the UK (central estimate, medium emissions scenario, 2080s).

(a) **(b)**

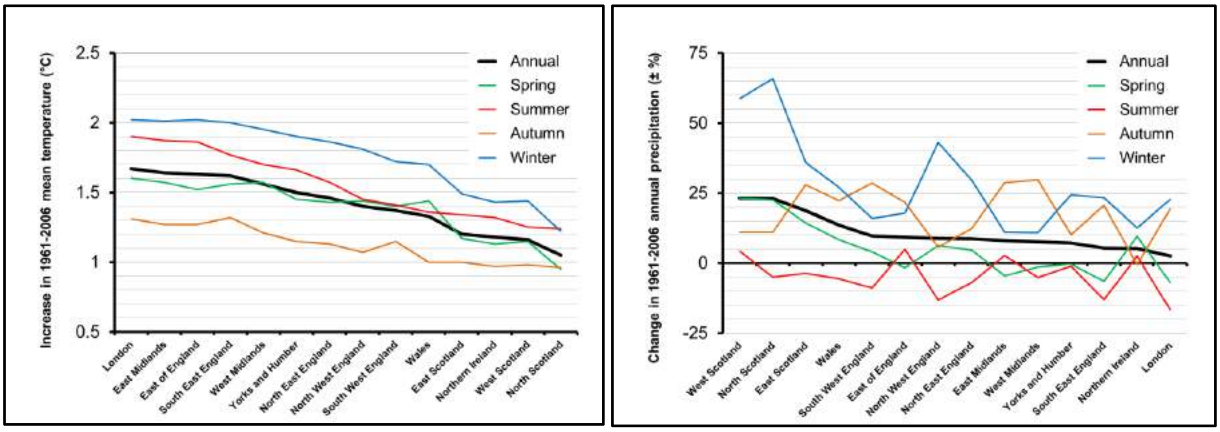


Figure 1-3 The change (1961-2006) in mean air temperatures (a) and annual precipitation (b) across the UK by region

Regions appear left to right in rank order of annual mean temperature rise. (Data UKCP09, reproduced with permission).

Extreme events

1.1.15 **Temperature extremes**: In addition to increases in mean temperature, the intensity and frequency of extreme events has shown some evidence of increase in the UK, with a suggestion that the warmest daily temperature extremes are rising faster than the rate of increase in mean temperature (UKMO 2014). The summer of 2003 is likely to have been the hottest in Europe for 500 years; this event was made twice as likely given the warming experienced up to that point (Jenkins et al. 2008, Stott et al. 2004). Given the 0.8 °C increase in mean summer temperature observed between 2003 and 2012, such an event is now ten times more likely (Christidis et al. 2015). The spatial gradient in mean temperature (most warming in the South of England, and least in the North of Scotland) is broadly replicated in the UKCP09 projections for maximum and minimum temperatures (Murphy et al. 2009).

1.1.16 **Precipitation extremes**: Precipitation has become more intense in winter throughout the UK and less intense in summer for all regions except for the North East of England and the North of Scotland (Jenkins et al. 2008). A UK-focused analysis has also suggested that extreme rainfall events in spring and autumn have increased, while longer duration rainfall events in winter have become more intense (Jones et al. 2013). In summer, shorter duration events have declined in intensity, while longer duration events have become more intense. As with the uncertainty over future projections of overall precipitation change, there is a substantial amount of disagreement between climate models with respect to future projections of changes in precipitation extremes. However, the UK Met Office has recently delivered a high resolution (1.5 km) UK-focussed model ('UKV') that represents the intensity and duration of rainfall events on a par with rainfall radar observations. This new model projected increases in hourly rainfall intensities, particularly in winter (by 2100, RCP 8.5 scenario, Kendon et al. 2014). Consequently there is a heightened flood risk for those catchments already receiving substantial rainfall in winter. In a departure from previous, coarser-scale projections, it also projected an intensification of short-duration rain in summer (Kendon et al. 2014).

1.1.17 **Drought**: For natural capital assessments, the availability of soil moisture (as 'stock', Robinson et al. 2009) is critical. Scientists work to the definition of drought as a 'sustained and regionally extensive occurrence of below average water availability' (Tallaksen and van Lanen, 2004, Hannaford et al. 2011). There are two means of classifying 'below average water availability': 1) meteorological, based on rainfall deficiencies, and 2) hydrological, based on runoff deficiencies. By these definitions, notable droughts occurred in the UK in the years 1975-76, 1984, 1990, 1995-97, 2003, 2004-06 and 2010-12. These droughts were notable for different reasons, with key differences in their impacts across regions and sectors of the economy (Marsh et al. 2007, 2013). Although summer (meteorological) drought events are projected to increase with climate change (UKMO 2012, and CCRA2 Evidence Project D), the evidence on changes to their observed frequency is equivocal (Watts et al. 2013).

1.1.18 To assess the worst case scenario for drought, the CCRA2 Evidence Project D derived a 'High++' scenario for low rainfall which would see summer rainfall deficits running at 60%, or 20% for longer durations (multi-annual). The impact of these events would be comparable to the worst drought events on record (1975-76 for short duration, 2010-12 for multi-annual, Wade et al. 2015).

1.1.19 **Storm surges**: Data from the UK Tide gauge network can be used to estimate the frequency and magnitude of storm surge events. These data show that extreme high (99%) sea levels at Newlyn and Aberdeen have increased broadly in line with the mean sea level rise for the UK (see below, Jenkins et al. 2008).

1.1.20 Future projected changes in storm surge events can be produced that either: a) account for anticipated rises in relative mean sea level, or b) do not account for sea level change. Without accounting for sea level change, UKCP09 projected that storm surges would not increase by more than 9 cm anywhere in the UK over the course of this century (Lowe et al. 2009). Estimates that do account for future sea level rise are subject to greater uncertainty; for 2100, the central estimate of surge heights with a 50 year return period is 0.4 - 0.5m higher than the present day baseline (Lowe et al. 2009).

Sea level

1.1.21 Overall sea level rise for the UK has been estimated at 1 mm yr^{-1} for the 20th century (Woodworth et al. 1999, Jenkins et al. 2008). A recent study estimated that global, present-day (1993-2010) sea level rise has accelerated to 3.0 ±0.7 mm yr^{-1} (Hay et al. 2015). Given that thermal expansion and loss of ice mass have been identified as the main drivers of this trend, the underlying rise in sea level globally will continue this century; however, there is less confidence in estimates of its future magnitude, with IPCC concluding that it is merely 'likely' (medium confidence) to be in the all scenario range of 0.26 – 0.82 m (IPCC 2013).

1.1.22 At a local level, vertical land motion can act to amplify the global trend, or offset it. In the UK, vertical uplift is occurring in North West Britain, while compaction and abstraction is lowering land levels in the South East of England. Past assessments of sea level change on particular regions or countries of interest (including the UK) have combined vertical land motion data with projections of sea level rise. These assessments could be improved by accounting for additional drivers of land-level change uncaptured by the vertical land motion data. These drivers include changes to ocean loading affecting levels in South West England, or the retraction of Fennoscandinavian ice mass affecting the east coast of Britain (Shennan et al. 2012).

Report structure

1.1.23 The remainder of this report is structured as follows:

- Chapter 2 focuses on impacts on the provision of clean water
- Chapter 3 focuses on impacts on equable climate
- Chapter 4 focuses on impacts on wildlife

Clean water

2 Clean water

Scope

2.1.1 In this review, we provide an overview of the impacts of climate change on clean water provision. The scope of the report is limited to impacts that: a) adversely affect clean water provision, and b) are increasing or are projected to increase in severity under future climate change.

2.1.2 We begin the chapter by establishing a baseline for the present day condition of the water environment across the UK using nationally consistent datasets (where these were available, Section 2.2). We then describe a number of risks to water quality with specific reference to the EU Directives to which they are relevant. Finally, we provide a summary of evidence gaps and conclude by providing a synthesis of overarching themes. The Appendix contains more information on the data-driven aspects of this review, including definitions of terms and the numbers behind the figures provided. Throughout, we treat 'clean water provision' as a final ecosystem service (as per the National Ecosystem Assessment 2011). This service leads to goods that are valued by the UK public, namely: drinking water; and water suitable for bathing and recreational activities such as fishing and watersports.

2.1.3 Our approach was systematic wherever possible. However, we would emphasise that it is not yet possible to make a systematic assessment of the relative importance of the risks posed by climate change to the delivery of water management objectives (cf. Arnell et al. 2014). We assembled evidence by meta-searching on Web of Science and Google Scholar, using search protocols based on the risks to clean water provision identified by the project team. A description of gaps in the evidence we found, and a discussion of why these gaps might exist, is provided in Section 0.

Definitions of impact, effect, hazard and risk

2.1.4 The words 'effect' and 'impact' are often used interchangeably in the scientific literature, although the latter is usually taken to mean an 'effect' that is marked, or notable. A 'hazard' is taken to be an agent of potential harm or damage, while (in formal terms) risk is a function of the magnitude of an effect and its probability. In this review, we followed the definitions of 'impact', 'hazard' and 'risk' set out in Humphrey (2014) as closely as possible. We do not use 'risk' to describe an 'effect' unless it had been assigned a probability. We did not consider all possible effects; rather, we restricted our scope to effects that have a notable adverse influence on society. Consequently the term 'impact' is used rather than 'effect'.

2.1.5 Over time, various definitions of all these words have been proposed in the academic literature, and more widely. On the rare occasion that the authors of a particular study have used a word or phrase differently to these conventions, an attempt was been made to 'translate' the meaning of the assertion into the terminology specified here.

Studies of climate change impacts on hydrology

2.1.6 The Centre for Ecology and Hydrology's Future Flows and Groundwater Levels (FFGWL) project was the first attempt to assess the impacts of climate change on hydrology in a nationally consistent manner (Prudhomme et al. 2012). FFGWL corrected for known issues with the use of raw climate model data, specifically regarding appropriate downscaling and the introduction of biases, such as the 'drizzle effect' of too much low intensity rainfall (Newton et al. 2012). It also operated at detailed temporal and spatial resolutions, the latter being particularly important for analyses of groundwater supply and flooding. It did not, however, account for land use change, which is, and will remain, a key modifier of hydrological regimes.

2.1.7 CCRA Project B built on the data and models behind the FFGWL project to generate projections of water availability for the UK. These projections were then compared to existing adaptation options, to identify where these options are at the greatest risk of being exhausted. Under high levels of climate and population change, the projections revealed that most of the UK (particularly South East England) would be in supply-demand deficit by the 2050s. The project also found that, under low levels of climate and population change, significant deficits would still be evident. This highlights that adaptations are required in certain areas of the UK irrespective of assumptions over future climate and population change.

Present day condition of the water environment

2.1.8 **It is critical to account for the current condition of any natural asset when assessing its vulnerability to climate change**. In this chapter, we establish the baseline condition of the present day water environment, and identify the existing risks to clean water provision.

2.1.9 EU Water Framework Directive data are the main source of information for this section; however, we also cover data collected under the requirements of the EU Bathing Water, Drinking Water and Urban Waste Water Treatment Directives.

Water Framework Directive data

2.1.10 To establish the present day condition of the water environment, we present data collected under the requirements of the EU Water Framework Directive (WFD; EU 2000, 2008). Because the remit of the WFD extends from source to sea, these data highlight spatial differences in the present day condition of the 'whole' water environment. They also provide a coordinated and integrated means of assessing the risk to a range of biological and chemical indicators of water condition. The WFD data thus support the trend assessments for the risks we identify in the following section.

2.1.11 Although the requirements for each EU Member State under the WFD are the same, the devolution of environmental powers within the UK means that the WFD data are summarised, and in some cases handled, differently. Thus we proceed with quantifying the drivers of these water quality data by taking each country in turn. Further detail on these drivers of water quality, and the role of climate change, can be found in Section 0.

2.1.12 **In England and Wales, 3,567 water bodies (46%) are predicted as a 'fail' under WFD for 2015** (Figure 2-1). The 'one-out-all-out' rule means that if a water body 'fails' (classified as less than good) under one WFD criterion, no matter its performance in the other criteria, it cannot achieve 'good' or 'high' status. This headline figure must not therefore be interpreted as an indication of status across all the WFD criteria.

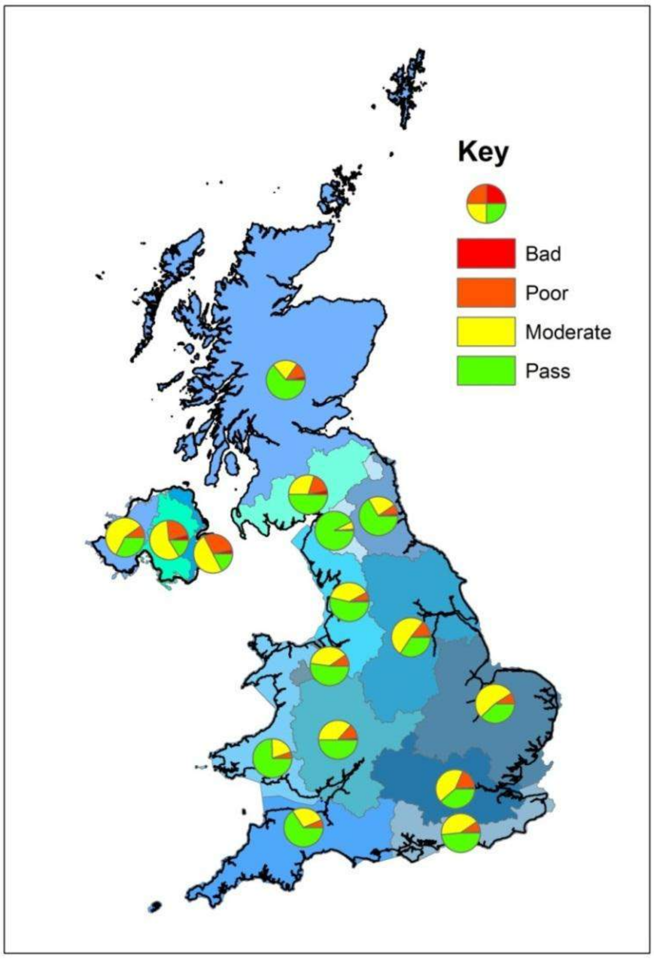


Figure 2-1 Summary of the overall status of UK water bodies as reported under EU Water Framework Directive (EU 2000, 2008) requirements

Pies indicate the proportion of water bodies in each RBD classified as 'bad', 'poor', 'moderate' or 'pass' (i.e. 'good' or 'high'). Data were derived from: the Environment Agency's data for England and Wales (WFD Cycle 2: predicted 2015 classification data); SEPA's WFD Cycle 2 online consultation tool (2013 data, excludes groundwater); and NIEA 2014 data.

2.1.13 The Environment Agency maintains a dataset documenting the water bodies in England and Wales that do not achieve at least a 'good' classification status under the WFD, and the reason why (the 'Reasons for not achieving good' dataset). Here, we present a summary of the numbers behind the version dated 14th August 2014. This version contains information on 3,919 of the 7,829 water bodies in England and Wales (including the Western Wales River Basin District, RBD). 'Certainty' in the attribution of SWMIs (Significant Water Management Issues) has increased since the beginning of WFD Cycle 1 (Figure 2-2). There has thus been increasing clarity as to the sectors (Figure 2-3), pressures (Figure 2-4 a) and sources (Figure 2-4 b) behind the failures reported. Definitions of all the terms used in these figures and further details of their derivation are provided in Appendix A.

2.1.14 Any particular water body may face multiple SWMIs that would have to be resolved for it to achieve a status of 'good' or 'high', (i.e. a pass), and more than one sector may be responsible for failures in each water body. Note that for 7,296 of the 13,929 reported failures, the specific pressure on the water body was not identified: thus, for over half the failures identified, the cause was unknown. Figure 2-4 b illustrates the sources of these pressures, where they are known. It is important to note that any remedial actions to resolve the issues presented in this dataset may already be underway, or in fact already completed. The beneficial consequences of these actions may not yet be apparent in the data. We therefore present these data as an indicative idea of risks to the present day water environment.

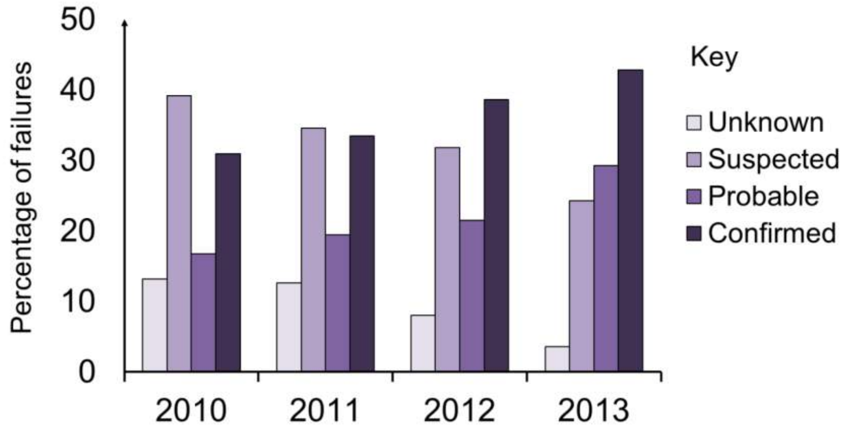


Figure 2-2 The change in certainty of WFD failure attributions in the Environment Agency's 'Reasons for not achieving good' dataset for England and Wales

2.1.15 **In Scotland, 1,225 (38%) of water bodies were reported as a 'fail' in 2013** (Data: SEPA WFD Cycle 2 online consultation tool). The Scottish Environment Protection Agency (SEPA) maintains a dataset similar to the 'Reasons for not achieving good' dataset, and these data were made available to the Project C team (Christie 2015). This dataset identifies the pressure type (e.g. abstraction), industry sector responsible (e.g. arable farming), and the relevant assessment category and parameter (e.g. water flow and levels; change from natural flow conditions) for 3,779 failures across Scotland (Scotland and Solway Tweed RBDs). Although the information on pressure type and source is recorded differently, it is clear that many of the reasons behind the failures are similar to those facing England and Wales (Figure 2-4a and Figure 2-4b).

2.1.16 The Northern Ireland Environment Agency (NIEA) does not maintain a dataset similar to the EA's 'Reasons for not achieving good' dataset, or the SEPA data. However, the NIEA do provide a written commentary on why some waterbodies within each RBD have not achieved a 'good' or 'high' status within the draft Cycle 2 River Basin Management Plans (NIEA 2014a,b,c), entitled: 'Significant issues and pressures'. All highlight the following:

- Failures in diatoms, macrophytes and soluble reactive phosphorous measurements (20-50% of water bodies across the three RBDs); all are indicators of possible nutrient enrichment from agricultural sources, sewage works, industrial discharges or septic tanks.
- Failures in invertebrates, dissolved oxygen and ammonia elements (20-50% of water bodies); these also indicate pressures from agriculture and points sources such as farms, sewage works, industrial discharges or septic tanks.
- Failures in fish (more than 10% in each RBD), indicating a wide range of pressures such as physical modifications, over-abstraction and factors affecting flow regulation.

(a) Sectoral provenance of failures in England and Wales.

(b) Sectoral provenance of failures in Scotland.

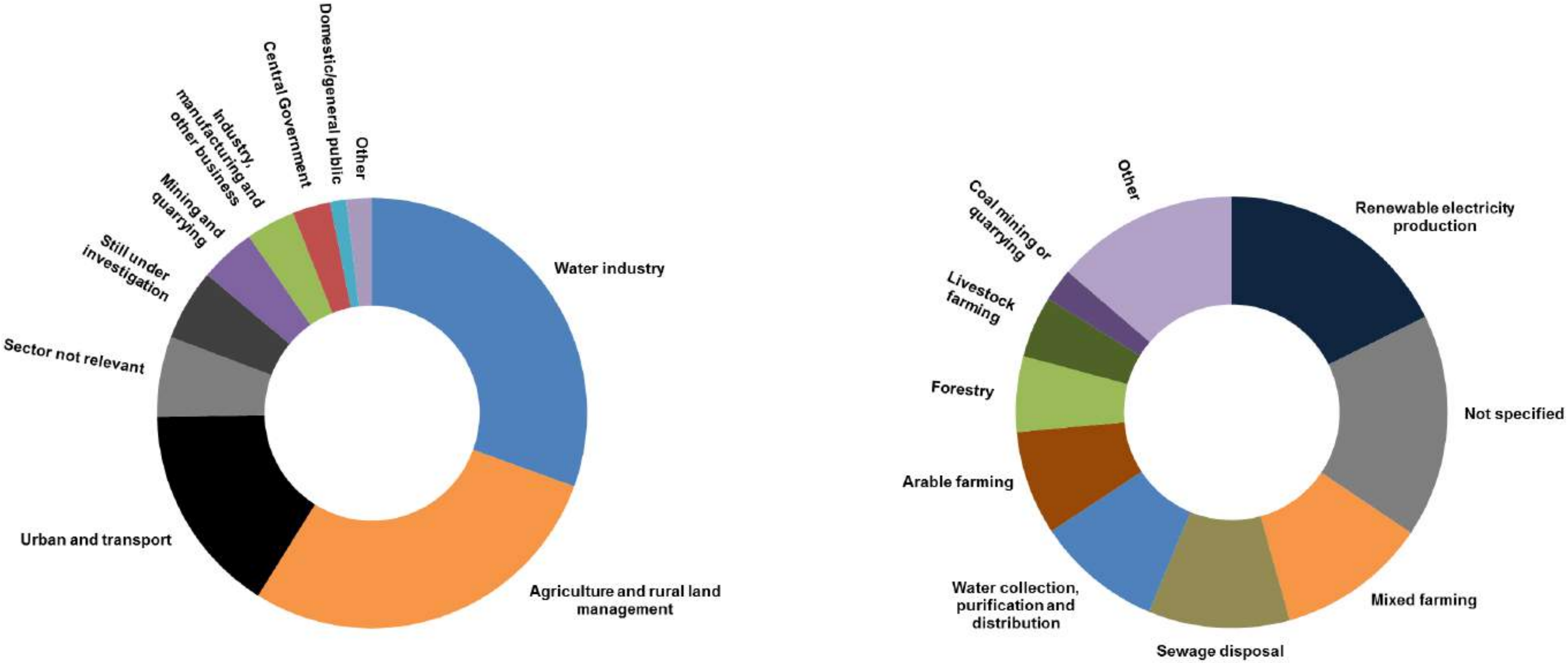


Figure 2-3 The proportional share of WFD failures attributed by sector: (a) in England and Wales, and (b) in Scotland

(Source: (a) Environment Agency 'Reasons for not achieving good' dataset, and (b) Christie 2015).

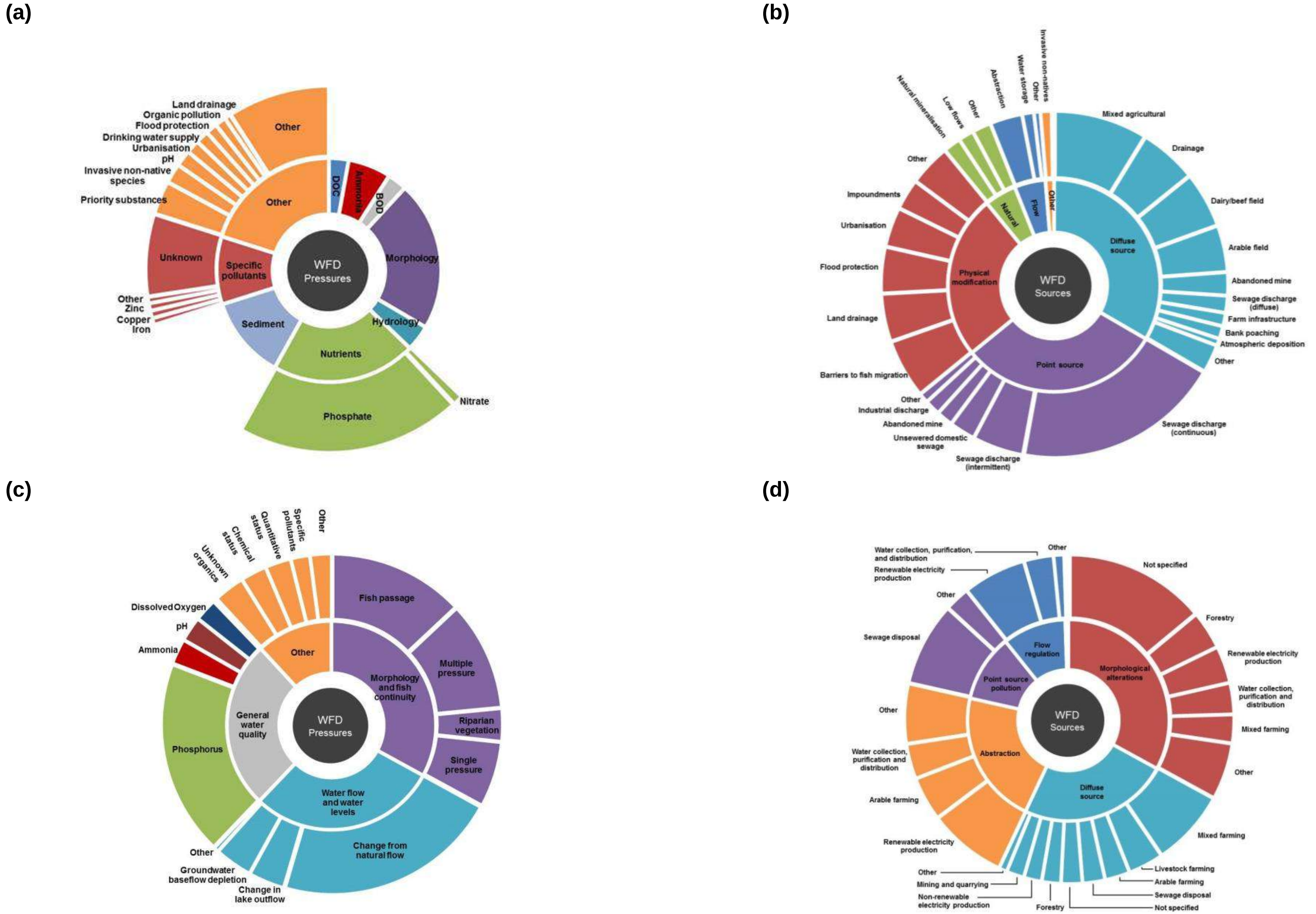


Figure 2-4 Pressures (a,c) and sources of pressure (b,d) identified as the cause of the WFD failures

(Source: Environment Agency's 'Reasons for not achieving good' dataset for England and Wales (a,b) and by SEPA in Scotland (c,d). In (a) and (c), the category of pressure appears in the inner circle, sized proportionately according to their frequency of occurrence as an SWMI. Where more information on the nature of these pressures is provided, this appears in the outer circle. In (b) and (d) the source appears in the inner circle, with more detail appearing in the outer circle..

Bathing Water Directive data

2.1.17 Monitoring under the Bathing Water Directive (EU 2006a) suggests that water quality at selected sites around the UK coast is improving. In England, compliance with the Bathing Water Directive has steadily improved from 65% of sites in 1988 to 99.5% today (2014). In 2014, compliance in Wales, Scotland and Northern Ireland was 100%, 98%, and 96% respectively (NRW 2014, SEPA 2015, NIEA 2015). A revised and strengthened Bathing Water Directive came into force in January 2015, and thus percentage compliance is likely to fall in the 2015 results. Nevertheless, anticipated performance under the new, more rigorous regime can be estimated for the historic data, and these estimates also show improvements with time (Environment Agency 2015).

Drinking Water Directive data

2.1.18 Drinking water quality is monitored under the EU Drinking Water Directive (EU 1998), and this monitoring suggests that the quality of drinking water has improved substantially over the last 25 years. This monitoring is conducted at the tap, water treatment works outflows, storage reservoirs, and water towers. Across the whole UK, the percentage of tests failing the standards at the tap has decreased from around 1.5 – 2% in the early 1990s to 0.1% today (DWI 2015a, b). Testing results at water treatment works, storage reservoirs and towers have also improved; for example, in 1990 there were 283 failures of tests for the harmful micro-organism *E. coli* across 174 treatment works in England (compliance 99.86%), yet by 2014 there were only three failures at three different works (DWI 2015a).

Urban Waste Water Treatment Directive data

2.1.19 The fate of wastewater and sewage is the subject of the EU Urban Waste Water Treatment Directive (hereafter the 'WWTD', EU 1991a). There are two products of wastewater and sewage treatment: 1) treated water that is released into the environment (and thus covered by WFD monitoring, see above) and 2) sewage sludge (Defra 2012). The WWTD required that sewage sludge discharges to surface waters cease by the end of 1998; these discharges were responsible for 281,588 dry tonnes of sewage sludge being released into the water environment in the baseline year of 1992 (Defra 2012). Higher standards of water treatment have also resulted in more sewage sludge being generated, such that the total weight produced by the UK per annum increased by ~ 50% between 1992 and 2010 (Defra 2012). This increase has largely been absorbed by agriculture (~ 80% of total UK sewage sludge reuse or disposal in 2010) although 18% is subject to incineration (Defra, 2012).

Risks to the provision of clean water

2.1.20 We proceed with a description of the risks to the provision of clean water which either: 1) have an increasing trend in the present day, which might be attributable to factors other than climate change; or 2) are projected to increase under future climate change. These criteria exclude some risks (e.g. acidification, direct alterations to morphology) with improving or ameliorating trends. In these cases, a substantial amount of existing information has already been published, for example, the detailed interpretive reports published by the Upland Waters Monitoring Network every five years covering acidification (among other impacts, Kernan et al. 2010), and synopses elsewhere in the literature (e.g. Monteith et al. 2014).

2.1.21 A number of the risks we identify are cross-cutting, being relevant to multiple water quality-related EU Directives. Most also have multiple causes and effects. For example, nutrient enrichment can begin with diffuse pollution in headwater streams, can be exacerbated by point sources of pollution on rivers, and can also be apparent in CSO (Combined Sewer Outflow) discharges to the sea. It therefore affects all the Directives referred to above. The section is thus structured by risk rather than by Directive. The relevance of each risk to each of the EU water-related Directives is shown in Table 2-1.

Table 2-1 The risks identified and their relevance to EU Directives

Risk	EU Directives			
	Water Framework Directive	Drinking Water Directive	Urban Wastewater Treatment Directive	Bathing Water Directive
Nutrient enrichment and eutrophication	■	■	■	■
Combined sewer overflows	■		■	■
Dissolved Organic Carbon	■	■		
Specific pollutants, priority substances, and 'other' chemical pollutants	■	■	■	■
Over-abstraction and saline intrusion	■	■		

Green colouring indicates that a particular risk falls within the remit of the Directives shown.

2.1.22 Each risk is broken-down into four constituent sections:

- a description of the problem in which the causes and mechanisms are discussed;
- the trend in the risk, assessed simply as increasing (↑), decreasing (↓), or stable (—),followed by a qualitative judgment of confidence in these trends ('high', 'medium', 'low'), as per the CCRA definitions (Table 2-2);
- the impacts of climate change on the risk; and
- adaptation measures to reduce the risk.

2.1.23 We also include four valuation case studies which provide an indication of the nature and significance of the costs and benefits associated with each of these risks, where this information was available.

Table 2-2 Definitions for the confidence levels used in the 'Trends' sections of this report

Confidence level	Definition
Low	Varying amounts and/or quality of evidence and/or little agreement between experts.
Medium	Several sources of high quality independent evidence, with some degree of agreement between studies, and/or widespread agreement between experts.
High	Multiple sources of independent evidence based on reliable analysis and methods, with widespread agreement between studies and experts.

Nutrient enrichment, and eutrophication

Description of the problem

2.1.24 Nutrient enrichment is still considered one of the biggest threats to meeting water quality objectives in the UK. Nutrients enter the water environment via a number of means, for example via domestic sewage or through fertiliser runoff. Eutrophication is the environment's response to excess nutrients, and often takes the form of excessive growth of plants or algae. This can lead to oxygen depletion, changes in plant community composition, increased presence of toxic algae, and a decline in biodiversity.

2.1.25 Typically, freshwater ecosystems are phosphorous-limited, while saline ecosystems are nitrogen-limited. Brackish estuarine waters may be limited by either or both, depending on conditions. Where the nutrient supply to an ecosystem is limited, nutrient increases typically result in increased productivity. This increase is more common in those organisms best positioned or adapted to take advantage of the additional nutrients, such as phytoplankton (or algae). The sudden, rapid spread of these organisms is known as a ‘bloom’. Plant species more adept at competition will thrive at the expense of others (especially macrophytes), often leading to a decline in other biodiversity. Indirect impacts can include a reduction in available oxygen, and an increase in turbidity (cloudiness or haziness). Reductions in oxygen can also facilitate phosphorous release from river beds, creating a positive feedback (see also Figure 2-15).

2.1.26 Oxygen depletion can result when organic matter is deposited into the water environment, enhancing microbial activity and therefore respiration rates. This organic matter can come from sewage and agriculture (silage and slurry), although in areas with historic pollution it can also be slowly released from the sediment bed. This can result in hypoxia or the death of organisms in the water, while less severe responses in fish include: avoidance responses (e.g. by migratory salmon); slower development and/or later hatching times; drops in migratory ability (such as maximum sustainable swim speed); and reductions in digestion efficiency (Carter 2005).

2.1.27 The discharge of agricultural runoff and sewage can also lead to an increase in the levels of nutrients and bacteria in bathing waters (Table 2-3). The ingestion of contaminated seawater by people can lead to diseases of the gastro-intestinal tract caused by bacteria, protozoans, viruses, blue/green algae and dino-flagellates, as well as respiratory and other problems. In addition, contact with polluted seawater can result in ear, eye and skin infections, and respiratory diseases (Georgiou and Langford 2002).

Table 2-3 Sources of faecal pollution within UK coastal bathing waters

Source	Range of Inputs
Continuous discharges	Wastewater treatment works Industrial discharges Cross connections to surface water outfalls
Intermittent discharges	Direct combined sewer overflows Storm-water overflows
Diffuse discharges	Riverine wastewater treatment works Riverine combined sewer overflows Unspecified urban diffuse sources Agricultural drainage and run-off septic tanks
Mammals and birds	Seal and bird colonies, dogs, donkeys
Harbours and marinas	Unsewered wastewater from boats

(Source: Georgiou and Langford 2002)

2.1.28 With respect to nutrients, studies suggest that the majority of nitrate loadings and almost half of phosphate loadings into coastal waters are attributable to agriculture within the UK (OECD 2012). With regard to bacteria, a study of water quality in the Ribble Estuary found that over 90% of the total bacterial organism load was discharged by sewage related sources during high flow events (Stapleton et al. 2008). The majority of this load was attributed to two waste water treatment plants, while diffuse sources included direct livestock defecation into watercourses, runoff of faecal matter from grazing areas, and/or runoff from areas receiving manure applications during periods of rainfall (Stapleton et al. 2008).

Trends in nutrient enrichment

Present day trend: ─ (medium confidence).

Future trend: ↑ (low confidence).

Trends in phosphate and phosphorous compounds

2.1.29 While phosphate features prominently amongst the causes of WFD failures in Britain (Figure 2.4a,c), phosphate inputs to the water environment have fallen. The Environment Agency (2012a) recently estimated that just under half the river length in England (and 6% in Wales) fails WFD standards for phosphorous. Entry into the environment is via two main anthropogenic sources: 1) point source pollution, primarily from sewage or industry (51% of phosphate failures in England and Wales), and 2) diffuse pollution, primarily from agriculture (47% of phosphate failures in England and Wales). The total weight of phosphate fertiliser applied to arable land has fallen by approximately 30% since 1990 (Johnston and Dawson 2005). This decline is due to the global shortage of rock phosphorous, and its impact on price (Soil Association 2010). Farmers have tended to use organic manure (not necessarily farmland manure) as an alternative (AIC 2014). Yields from arable crops (particularly cereals and oilseeds) are strongly linked to the availability of phosphorous in the soil, i.e. they are phosphorous-limited. Thus, diffuse pollution of phosphate is likely to remain a concern in agricultural areas, particularly where the water table is shallow.

2.1.30 Detergents (mostly sodium tripolyphosphate, STPP) are another source of phosphate which is released into water from sewage and industrial point sources. This is because the phosphate ‘softens’ the water by removing calcium and magnesium. Controls have recently been introduced to limit phosphate content in detergents that may already be reducing entry into the water environment via this means (see ‘Adaptation’ below). The group responsible for devising and reviewing WFD standards (UK technical advisory group, UKTAG) has proposed revised, more stringent standards for phosphorous (UKTAG 2013a) that account for site alkalinity and altitude, and better reflect its impact on biology. It is estimated that the adoption of these new standards in the next WFD cycle would result in a 15% increase in WFD failures (UKTAG 2013a).

Trends in nitrate and nitrogen compounds

2.1.31 Nitrogen causes fewer failures under the WFD, and entry into the water environment is primarily via diffuse sources. Entry can be via its use as a fertiliser in agriculture or from sources in the wider landscape, such as upland areas affected by atmospheric nitrogen deposition. Use of nitrogen fertiliser in grassland has declined by 50% since the 1990s, yet use in arable farmland has remained relatively constant during this time (AIC 2014). The decline in nitrogen use on grassland has been attributed to a drop in total cattle numbers (a decrease of 13% between 2000 and 2014), and improved use efficiency of manure (Defra 2015a). The most common form of agricultural nitrogen fertiliser is ammonium nitrate. Thus, as with manure, its use is reflected in levels of ammonia in the environment. Ammonia is responsible for 6.2% of WFD failures in England and Wales, although it is primarily monitored for its toxicity to fish. Under aerobic conditions it is volatilised by nitrifying bacteria to nitrite, then to nitrate (Environment Agency 2007). Nitrate is responsible for 0.4% of WFD failures in England and Wales.

2.1.32 Entry to the water environment via atmospheric release is typically due to fossil fuel combustion (as oxides of nitrogen, or NO_x), and occurs during power generation, industrial processes or transportation. Once in the atmosphere, NO_x is converted into nitric acid (HNO_3), which is deposited on land or directly into surface water. This deposition can be dry or wet (more commonly known as acid rain). Emissions of nitrogen to the atmosphere are in decline across the UK, particularly from power stations and road transport, which have been subject to stringent emissions control (Rotap 2012). Furthermore, ecosystems badly affected by nitrogen deposition are showing signs of recovery (Kernan et al. 2010).

Impacts of climate change

2.1.33 The principal means of nutrient entry to the environment involve direct or indirect human activities, and thus the impacts of climate change exert less of a control over nutrient enrichment than they do over other aspects of water quality. Nevertheless, climate change could play an important role in exacerbating existing (natural and anthropogenic) drivers of nutrient enrichment (Table 2-4). In this context, changes in the seasonality of precipitation will be critical. There will be less dilution of nutrients in summer if projected changes to precipitation are accurate: this is the season when dilution is required the most. The possible impact on dilution would be of most concern at point sources (Arnell et al. 2015). Paradoxically, potential winter precipitation increases also pose a risk to nutrient status via increases in erosion and accelerated nutrient transport. Where summer droughts follow wet winters, algal blooms are more likely to occur (Paerl and Huisman 2008).

2.1.34 Higher temperatures will increase rates of production and decomposition in water bodies, thereby raising respiration rates in algae and lowering dissolved oxygen levels (Jeppesen et al. 2010). The risk of low dissolved oxygen during algal blooms will be more pronounced during low flow conditions, and at night (Williams et al. 2000, Cox and Whitehead 2009). The simultaneous impacts of higher temperatures, which directly cause fish stress, and reductions in available oxygen, could combine to cause fish mortality.

Table 2-4 Overview of climate change impacts on nutrient enrichment and eutrophication

Direction of impact	Climate variable		Mechanism(s)
	Present trend (confidence)	Future trend (confidence)	
Exacerbating	Temperature		Accelerated growth of algae; lower dissolved oxygen
	↑ (High)	↑ (High)	
	Winter precipitation		Soil erosion; nutrient cycling
	↑ (Low)	↑ (Low)	
	Summer precipitation		Less dilution; longer residence times
	↓ (Low)	↓ (Medium)	
	Drought		Increased mineralisation
	– (Low)	↑ (Low)	
	Storminess		Increased Combined Sewer Overflows (CSOs); increased nutrient load from agriculture
	- (High)	↑ (Medium)	
Mitigating	Annual precipitation		Reduced runoff, longer residence times
	– (Low)	↓ (Low)	

(The present and future trend for each climate variable is provided, a description of the mechanism involved, and the direction of the impact on nutrient enrichment (exacerbating or mitigating).

2.1.35 Projected decreases in overall precipitation could also act to reduce the amount of runoff, thereby increasing nutrient residence times in the soil. This would result in reduced loss of phosphorous from agricultural land. A recent climate modelling study for England and Wales found that annual losses of phosphorous from agriculture would decline by 4% by 2030, and 16% by 2050, under a medium emissions scenario (Cooper et al. 2010). This reduced runoff was particularly evident in the South and East of England, where the dominant flow pathway to rivers is artificial drains. The link between reduced runoff and reduced loss of phosphorous was confirmed in a similar climate/land use modelling study (Crossman et al. 2013), although the climate model applied in this latter study (KNMI RACMO, Medium SRES A1b scenario) projected an increase in annual precipitation overall.

2.1.36 Climate change could also change the frequency and type of algal blooms typically experienced in the UK. Algal species that are more tolerant of hotter temperatures are becoming more widespread and more dominant in their existing habitats (Paerl and Paul 2012). Cyanobacteria were first noted as a national concern for the UK following the widespread cyanobacterial blooms of autumn 1989, particularly in Rutland Water and Rudyard Lake (Krokowski and Jamieson 2002). Cyanobacteria prefer warmer temperatures, particularly over 25°C, and are already outcompeting diatoms and green algae in warmer summers. The competitive advantage of cyanobacteria is not only the result of faster growth rates, but also of the reduced vertical mixing that occurs at higher temperatures, which favours the highly buoyant cyanobacteria (Jöhnk et al. 2008). Many cyanobacteria are also more tolerant of low light levels typical of nutrified habitats. Climate change has also been implicated in the global spread of a prominent species of cyanobacteria (*Cylindrospermopsis raciborskii*) towards higher latitudes (Saker et al. 2003). This species was previously described as a tropical or sub-tropical species originating in South America, and has spread polewards after human introductions to other continents. Systematic monitoring for cyanobacteria, and eutrophication of all kinds, is expensive (Mainstone 2010), and therefore information on the identities of the species likely to spread and establish in the UK is lacking. However, it is likely that many of these species are already present in UK water bodies, and will establish more widely as they become more competitive due to climate change.

2.1.37 Although toxins produced by some species of cyanobacteria have been linked to a number of human health complaints (skin, liver, gut, and neurological system), the human health impacts of the most widespread toxins produced by cyanobacteria are not clear (e.g. β-methylamino-l-alanine or BMAA, Rumsby et al. 2008). Furthermore, the final trigger causing a cyanobacterial bloom (be it nutrification, a warm spell of temperature, or something else) can be difficult to pinpoint (Tyler et al. 2009). Thus it is difficult to assess the efficacy of existing water treatment techniques in mitigating the risk.

2.1.38 Projected increases in storminess will increase CSO discharges, and specifically the direct input of untreated sewage into the water environment. Aside from direct human health impacts on bathers or recreational users, this will increase the input of nutrients. Increases in winter precipitation would increase soil erosion and nutrient input (Knox et al. 2015). Given that such a precipitation increase will also increase flood risk and sediment delivery, the potential for a cascade of negative climate change impacts should not be underestimated (Lane et al. 2007).

Adaptation

2.1.39 There are a large number of measures already in place (or proposed) to tackle nutrient enrichment. Some of the key ones amongst these are described below, ordered by target area, from source to sea.

2.1.40 The prevention of diffuse pollution is underpinned by regulations at the EU level that cover both: a) upstream, land management issues (e.g. the EU Nitrates Directive 1991), and b) downstream impacts in the water environment (the EU WFD). Overall, there are a variety of preventative actions that could be implemented to reduce nutrient enrichment. Many of these need to be enacted at the catchment scale to be effective, i.e. the scale at which multiple landowners or managers can be engaged with a common goal in mind. To this end, Defra, the Environment Agency and others are promoting a catchment-based approach, with a focus on collaborative decision making, and an emphasis on 'upstream causes, downstream effects'. Managing at the catchment level involves both public and private stakeholders in the identification of issues which, if effectively addressed, would improve the provision of clean water. In England, the new Countryside Stewardship scheme will also provide capital grant funding in target areas (defined by the Catchment Sensitive Farming project) known for diffuse pollution (n=77 in 2015, administered by Natural England). In Wales, the new Environment (Wales) Bill will require measures to be outlined in 'area statements' at the river catchment scale (or at an alternative geographic scale that is more appropriate, Cowdy 2015). In Scotland, 14 priority catchments have identified by SEPA for targeted measures. The difficulties involved in diffuse pollution control are illustrated by the existence of Countryside Stewardship grants for 45 different types of farming infrastructure in England (Natural England 2015), under the new England Rural Development Plan (RDP). Similar commitments to grant funding will be outlined by the 2014-20 RDPs for Scotland and Wales.

2.1.41 The WFD (Article 7) requires Drinking Water Protected Areas (DrWPAs) to be designated for water bodies where drinking water is abstracted. Rivers, lakes, reservoirs and (all) groundwater supplies are included. Deterioration inside DrWPAs must be prevented if/where it would lead to additional treatment being required. Safeguard Zones can be designated if a DrWPA is found to be at risk, in which case careful management of problem substances is required. Due to the risk from nitrate pollution (and pesticides), large areas of Wales, South West England and the East of England are designated as Safeguard Zones for groundwater supplies. Surface water DrWPAs are generally designated for colouration and pesticide issues.

2.1.42 Advisory responses have also been developed to address diffuse pollution, as the associated land management issues often reoccur. For example, avoiding the application of fertiliser when the weather or soil conditions are likely to lead to runoff can mitigate impacts, as can tailoring fertiliser application to the needs of the crop. Both approaches are likely to be facilitated by technological developments in weather forecasting and so-called precision agriculture. Other guidelines, such as applying fertiliser in spring rather than winter, and avoiding grazing immediately after manure spreading, can reduce nutrient runoff of both nitrogen and phosphate (NFU 2009). Country-specific advice is provided by the Catchment Sensitive Farming initiative in England, Farming Connect in Wales, and SEARS in Scotland (Scotland's Environment and Rural Services). Reductions in pollution from runoff can also be driven from the bottom-up, with local groups (such as the Tamar Organics group in Cornwall) facilitating information and knowledge exchange. Groups such as these can empower farmers to make informed decisions based on authoritative, impartial advice (Global Food Security 2015).

2.1.43 Substantial reductions in nutrient runoff can also be achieved by the improved provision (and enforcement) of compliant septic or slurry tanks. Gains are especially apparent where the existing systems do not have adsorption beds or soakaways (Postma et al. 1992). Studies in Cumbria (Ockenden et al. 2014) and the cross-border region of Ireland (Blackwater River, Macintosh et al. 2011) have both demonstrated reductions in total phosphorous of approximately 25% downstream in the years following improvements to tank systems. The improvement at Blackwater River was also apparent during periods of low flow, in which the downstream ecology will be more sensitive to nutrient input.

2.1.44 Once nutrients have entered the soil via diffuse sources, the creation of riparian buffer strips can reduce onward transport to water bodies by slowing the velocity of surface and sub-surface runoff, and reducing the transport of suspended particles (Muscutt et al. 1993). Because the effectiveness of buffers depends on catchment characteristics and pollution levels, the degree of buffering provided is often variable (Stutter et al. 2012). However, evidence for the relative merits of different types of buffer is more established, with woodland strips (Nisbet et al. 2011) or heavy grasses (Lee et al. 2003) often cited as having the largest beneficial impact. Soft or 'green' engineering techniques, such as geotextile mesh or willow hurdles, are advocated as a means of facilitating the establishment of buffers where the soil is bare to begin with (SEPA 2009). Buffers can also benefit hydrological and ecological connectivity, reduce sediment loss and mitigate flood risk (Stutter et al. 2012). Buffers, by providing riparian shading, are also considered a useful adaptation to offset rises in stream temperature (cf. Lenane 2012).

2.1.45 Reduced transport of nutrients into water bodies can also be achieved through measures that decrease soil erosion. Standards for soil management are set out in Cross Compliance requirements under the Common Agricultural Policy.[1] Funding under basic payment, agri-environment and some RDP schemes is contingent on meeting these requirements. Good practice is also set out in codes of good agricultural practice that go beyond the minimum regulated standards (e.g. Defra 2009). Improvements are also driven by environmental stewardship schemes, such as Defra's Entry Level Stewardship Scheme in England (Defra 2005).

2.1.46 To target point source pollution from sewage works, the EU WWTD sets out the level of treatment that specific works must provide. This level of treatment is based on the 'population equivalent' they serve, and the sensitivity of the water body they discharge into. The staggered phasing of these requirements has delivered substantial reductions in the amounts of nitrogen and phosphorous discharged to water bodies across the UK. By treating with a precipitant, it is also possible to remove phosphate from the effluent at sewage works in a (tertiary treatment) process known as phosphate 'stripping'. This can have beneficial impacts even in water bodies with longer residence times, such as lakes (Parker and Maberly 2000). In other areas where phosphate has already permeated the sediment bed extensively, such as the Broads, suction dredging is deployed.

2.1.47 The EU has also regulated the use of phosphates and other phosphorous compounds (EU 2004, 2012) in industrial and domestic detergents, which would otherwise increase the need for wastewater treatment. The initial proposal for limiting the phosphate content of cleaning products was 0.4% by weight (Regulation 9); this was amended to 0.5 grams per recommended wash quantity. There are also mandatory requirements for biodegradability and labelling. Although the residence time for phosphate in the water environment is relatively short-lived (especially in rivers), it is probably too early to assess the impact of these (2012) legislative changes on phosphate levels in water. Many detergent manufacturers are considering the complete removal of phosphate from their products, and some have already done so.

2.1.48 Although mainly focused on organic content (primary and secondary treatment), the WWTD also includes provision for pollutants that remain after these first two levels of treatment, where the water body is: a) eutrophic, b) an abstraction source and high in nitrate, or c) at risk of failing under other Directives without the introduction (or continuing provision of) tertiary treatment. Such water bodies are designated 'sensitive areas'. Note that they have a different definition and scope to the similarly named 'Nitrate Sensitive Areas' of the 1990s, set up in accordance with the EU Nitrates Directive (EU 1991b). These were expanded into the larger suite of Nitrate Vulnerable Zones (NVZs) in 1996. This suite of NVZs is reviewed and expanded periodically, based on available evidence. Land is designated as within a NVZ if it is eutrophic, or if it drains into surface or groundwaters with 50mg of nitrate per litre (groundwater stores are particularly sensitive to nitrate pollution from agriculture). NVZs cover most of lowland Britain, especially England.

2.1.49 In water bodies which suffer from severe depletion of oxygen, the injection of pure oxygen is sometimes used. This deployment is via land injectors and oxygenation barges (the 'Water Witch') in Cardiff Bay, via barges on the River Thames (the 'Thames Bubbler' and 'Thames Vitality'), and via land injectors on the River Clyde. On the Clyde, which was previously classified as 'poor' for dissolved oxygen under the WFD, SEPA and Scottish Water installed two oxygen injection units at Scottish Water's Shieldhall wastewater treatment works, finding an improvement of 1-1.5 mg/litre in dissolved oxygen that extended more than 2km in each direction (Ellaway 2012). These measures are more likely to be taken up in catchments where changes to the morphology are either expensive or impossible, such as built up urban areas and/or heavily modified water bodies. Such schemes may also be appropriate where further improvements to wastewater treatment may be costly or unachievable for one reason or another.

[1] Cross Compliance is a mechanism that links direct payments to compliance by farmers with basic standards concerning the environment, food safety, animal and plant health and animal welfare, as well as the requirement of maintaining land in good agricultural and environmental condition. Since 2005, all farmers receiving direct payments are subject to compulsory cross-compliance (see http://ec.europa.eu/agriculture/envir/cross-compliance/index_en.htm).

2.1.50 The EU Bathing Water Directive sets parameters on the level of contaminants that can be present within bathing waters. Since the adoption of the Bathing Water Directive in 1976, there has been considerable improvement in the quality of bathing waters as a result of investments in water infrastructure (Georgiou and Langford 2002). This has meant that, although sewage pollution has been the major source of pressure on bathing water in recent times, other sources are becoming more dominant, particularly during wet weather (Georgiou and Langford 2002).

2.1.51 As such, while substantial improvements in bathing water have been secured as a result of the many investments in new sewage treatment schemes, further improvements are becoming increasingly difficult to deliver, largely due to the role of diffuse non-sewage sources of faecal bacterial pollution from catchments draining to the coast (Georgiou and Bateman 2005). A revised Bathing Water Directive was introduced in 2006 (and which is to be fully implemented by May 2016) with the aim of setting more stringent water quality standards and also putting a stronger emphasis on beach management and public information provision (EU 2006).

2.1.52 For CSOs, measures such as the provision of real-time monitoring data to bathers and recreational water users, or increasing the storm water capacity of the infrastructure can be deployed to mitigate the risk (See 2.1.95 for more detail on CSO adaptations).

Valuation case study #1: Nutrient enrichment in bathing waters

2.1.53 The link between climate change, bathing water quality and human wellbeing is set out in the impact pathway in Figure 2-5. This shows that an increase in certain types of bacteria in bathing waters could have a direct effect (shown by the dashed arrow) on human health while nutrient enrichment may stimulate the proliferation of cyanobacteria (algal blooms) which may produce toxins that are harmful to human health when swallowed, come into contact with skin or inhaled (WHO, 1999).

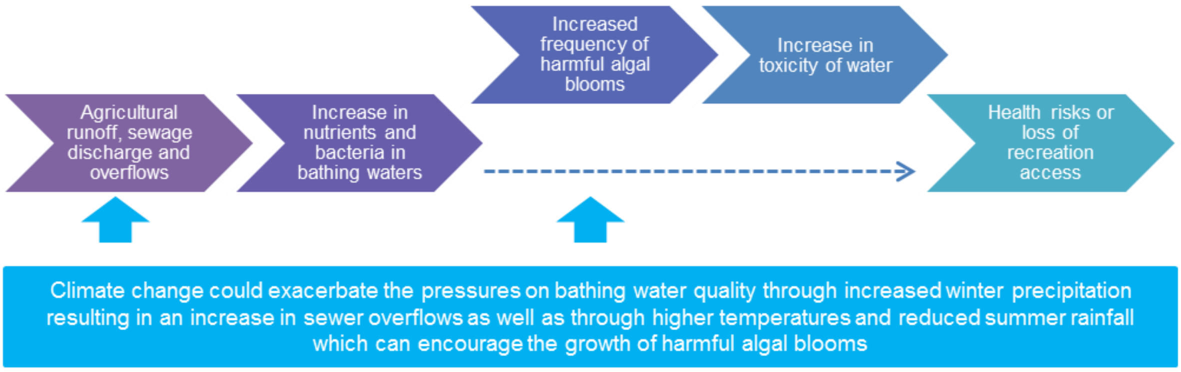


Figure 2-5 Impact pathway describing the impact of nutrient enrichment on human wellbeing

2.1.54 Although algal blooms are a natural part of the phytoplankton seasonal cycle, some can have a negative impact on marine ecosystems and the services they provide (such as recreation, fisheries, and shellfish production). These are termed ‘Harmful Algal Blooms’ or HABs. HABs can lead to the production of potent toxins, which can pose risks to human health and accumulate in filter feeding shellfish or other marine species. High biomass algal blooms can also cause water colouration, which has an unsightly appearance. HABs occur in regions in the UK with a strong Atlantic influence, although impacts are generally lower in Wales, Northern Ireland and the Isle of Man (Bresnan et al. 2013).

2.1.55 **The relationship between anthropogenic nutrient enrichment and HABs is complex**, and HABs can occur naturally in areas without pressures of nutrient enrichment. Loch Creran on the west coast of Scotland, for example, suffers HAB events despite relatively low nutrient concentrations and few anthropogenic inputs. Furthermore, while organic nutrients have been shown to support the growth of a range of HAB species, it is difficult to establish if these nutrients specifically promote the growth of harmful species (in preference to benign ones), or if they influence toxicity (Davidson et al. 2014).

2.1.56 Moreover, while the number of recorded HAB events has been increasing, there is no documented parallel increase in human and other animal health events. Although the adverse health effects from exposure to harmful algal toxins has been known for decades (for some of the cyanobacterial toxins), very few epidemiologic studies designed to systematically assess these effects have been undertaken. As noted by Moore et al. (2008) this has hindered the Intergovernmental Panel on Climate Change (IPCC) projections of climate change impacts on HAB-related illnesses (Confalonieri et al., 2007), and will also make the detection and quantification of climate change impacts on HAB-related illnesses difficult.

2.1.57 Because the link between anthropogenic nutrients and HAB events is not universal, it is difficult to point to HAB-related human health incidents that are the consequence of anthropogenic nutrient enrichment, although this is likely because most HAB observations result from HAB monitoring programs designed to ensure shellfish safety, not to take into account direct risks to human health (Davidson et al. 2014).

2.1.58 As such, it is difficult to quantify a relationship between nutrient levels in bathing waters and potentially negative impacts on human health and recreation. Nevertheless, Georgiou and Bateman (2005) outlined some of the minor morbidity effects that have economic consequences that could potentially arise from nutrient-enriched bathing waters. These included:

- **Medical and care-giving costs**, such as: out of pocket medical expenses of the affected individual (or family), the opportunity cost of time spent in obtaining treatment, plus costs paid for insurance, etc. Individuals may also be unable to undertake some or all normal chores and thus require additional special care-giving and services not reflected in normal medical costs.
- **Loss of time/productivity at work**. This includes personal income foregone, plus lost productivity irrespective of whether the individual is compensated or not (whilst some individuals may be paid sick pay and hence not perceive any income loss, sick pay is nevertheless a cost to businesses and in this respect reflects lost productivity).
- **Other social and economic costs**. These may include lost opportunities for enjoyment of leisure activities, discomfort or inconvenience (pain and suffering), anxiety, concern and inconvenience to family members and others. In addition, individuals may engage in defensive and averting expenditures and activities associated with attempts to prevent these health impacts.

2.1.59 Detailed economic evaluation and cost/benefit analysis of the impact of anthropogenically generated HABs is, however, lacking, since the economic impacts of HABs can be attributed to anthropogenic factors only if a causal link can be shown to exist between them. Davidson (2014) noted that much of the literature on the economic impacts of HABs has employed relatively crude measures and methodologies, the results of which often are difficult to compare. He therefore suggests that there is a need for more targeted research to understand the consequences of HAB events associated with changes in recreational activities, fishery closures, or health risks. While several studies of the economic impacts of HABs do exist internationally (see, for example, Anderson et al., 2000; Hoagland and Scatasta, 2006; Morgan et al., 2009 and Nunes and van den Bergh, 2004), these studies have not demonstrated clear links to anthropogenic nutrients and there is a paucity of studies looking at such issues in the UK (Davidson et al. 2014).

2.1.60 It is, however, possible to obtain an indication of the value of changes (an improvement or deterioration) in bathing water quality more generally based on studies of peoples' stated and revealed preferences for these changes. In this regard, several such studies were undertaken as part of efforts better to understand the costs and benefits associated with the introduction revisions to the Bathing Waters Directive.

2.1.61 While the original 1976 Bathing Water Directive led to significant improvements in bathing water quality across the UK, in 2002 a proposal was made to implement a number of changes to the 1976 Directive to reflect a combination of improved scientific knowledge, a recognised need for more active quality management of bathing waters, and a desire for improved public information provision. The main features of the proposals for the revised Directive were (Georgiou and Bateman 2005):

1. Bathing waters must meet ''good quality'' status, to be determined by meeting two microbiological parameters over three preceding seasons. This was tighter than the standard set by the 1976 Directive. Achievement of ''excellent quality'' was to be promoted. This was twice as stringent as ''good quality'' status.
2. Water quality should be monitored more frequently, albeit across a more restricted number of parameters.
3. Basic bathing water management measures should be introduced (which would improve knowledge of the risks and impacts) alongside improved provision of information to the public.

2.1.62 Mourato et al. (2003) undertook a study to reveal people's WTP (in England and Wales) for changes in water quality that were likely to arise from implementation of a revised EU Bathing Water Directive. Water quality was described in two different ways, giving rise to two versions of the questionnaire, A and B. In version A, water quality was described as the risk of suffering a stomach upset from bathing whereas, in version B, water quality was described as the number of days in the bathing season when it was considered unsafe to swim due to an elevated risk of suffering a stomach upset. The study found that respondents were willing to pay between £1.10 and £2.00 per year for a 1 in 100 reduction in the chance of contracting stomach illness from bathing water at a typical beach, and between £0.90 and £1.10 per year to avoid one day of poor water quality (Mourato et al. 2003).

2.1.63 Hanley et al. (2002) used a combined stated and revealed preference approach to value coastal water quality improvements, focusing on an area of Scotland that had consistently failed to meet standards under the Bathing Water Directive. The authors employed the travel cost method to estimate the change in the number of trips to a water resource if water quality improved and the welfare increase associated with each trip. The study found that a potential investment by West of Scotland Water that would guarantee that bathing water quality would meet EU standards at all sites during all sampling periods, would lead to an estimated increase in trip frequency of 1.3 per cent, generating a consumer surplus of £0.48 per trip or £5.81 per person per year.

2.1.64 A survey by Georgiou et al. (2000) involved interviewing members of the public at two seaside towns, Lowestoft and Great Yarmouth, and in Norwich, roughly 20 miles inland. People were asked whether or not they would be willing to pay higher water rates in order for all the beaches in the Anglian Water region to pass the new standard set by a proposed revision to the EU Bathing Water Directive. The average WTP was found to be £35.73 per household, roughly the same order of magnitude as the estimated potential cost increases in average annual household water bills necessary to implement the proposed improvement in bathing water quality. The exact terms of the new EC standard were not set out in the questionnaire as it was considered that the criteria for conforming to indicator bacteria counts would be meaningless to the public and it was not possible to give precise reductions in illness incidence (Georgiou et al. 2000). A summary of the results of the review of value estimates is set out in Table 2-5.

Table 2-5 Summary of the results of the review of value estimates

Value estimate	Aspect considered	Reference
£1.10-£2.00 per year (2002 prices)	WTP for a 1% reduction in the chance of contracting stomach illness from bathing water	Mourato et al. (2003)
£0.90-£1.10 (2002 prices)	WTP for a one day reduction in the number of days of poor water quality	Mourato et al. (2003)
£0.48 per trip or £5.81 per person per year (1999 prices)	Value of change in coastal water quality in Scotland from failing to meet the standards of the Bathing Water Directive to continually meeting the standards	Hanley et al. (2002)
£35.73 per household (1999 prices)	WTP to improve water quality at beaches in the Anglian Water region	Georgiou et al. (2000)

(Source: Georgiou et al. 2000.)

2.1.65 Georgiou and Bateman (2005) reviewed a number of studies looking at the costs and benefits of bathing water quality improvements to inform an analysis of the costs and benefits to the UK of proposed revisions to the 1976 Directive. A key finding was that the ratio of benefits to costs reported in the studies reviewed depended on the level of water quality and associated health risk reduction that compliance with Directive was expected to deliver, as well as the types of health and other benefits included in the assessment.

2.1.66 Since the studies reviewed differed in their approaches in terms of both defining changes in the degree of water quality improvements and the types of benefits to be experienced, Georgiou and Bateman (2005) found it difficult to accurately quantify the costs and benefits of bathing water quality improvements as a result of the Directive. As a result, the study developed a range of estimates for the potential costs and benefits of the Directive which varied significantly. For example, if it was assumed that the Directive met the most stringent of water quality scenarios for the UK, the costs were estimated to be £9,119 million (in 2002 prices). Comparing this to the broadest estimate of the potential benefits of £12,983 million (Georgiou et al. 2000), generated an expected benefit:cost ratio of 1.42. However, comparing this against the most conservative estimate of the benefits at around £450 million (Mourato et al. 2003) gave a benefit:cost ratio of 0.05.

2.1.67 On the basis of these results the authors concluded that, although not unequivocal, there appeared to be some support for the tightening of bathing water standards in terms of the economic costs and benefits associated with a revised Directive.

Combined Sewer Overflows (CSOs)

Description of the problem

2.1.68 Combined sewers convey both sewage and rainwater runoff to sewage treatment works for treatment prior to discharge. These differ from more modern separate sewers which transport only sewage, with surface water run-off being directed to its own dedicated drainage system. Combined sewer systems comprise ~40% of the total sewerage network in England and are designed with limited capacity for peak surface water flows (HM Government 2012).

2.1.69 Heavy or prolonged rainfall can rapidly increase the flow in a combined sewer until the volume becomes too much for the sewer to carry and excess storm sewage is discharged to rivers or the sea via combined sewer overflows (CSOs). A study of CSO discharges in the River Thames, for example, found a clear correlation between mean monthly rainfall and CSO discharges as set out in Figure 2-6.

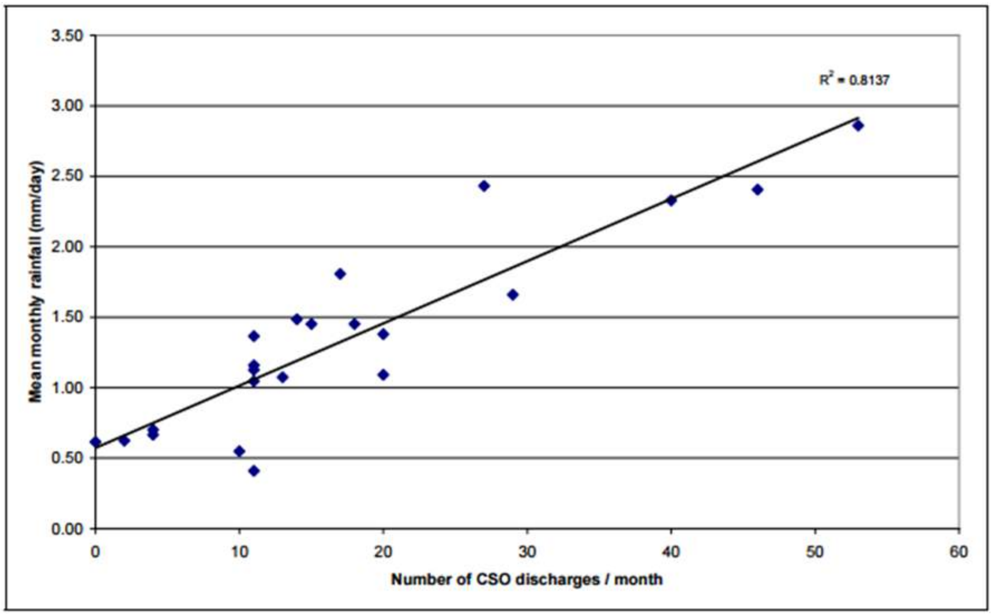


Figure 2-6 Correlation between the number of CSO discharges and the mean monthly rainfall in the Thames

(Source: Lane et al. 2007)

2.1.70 Well-designed CSOs act as an essential relief valve, preventing overloading which would otherwise lead to flooding of properties and sewage treatment works. However, many older CSOs were designed and constructed to a much lower standard that is now considered acceptable. During periods of intense rainfall, these become overwhelmed resulting in flooding of properties, gardens and public spaces and discharge untreated water into adjacent watercourses. These discharges can also result in oxygen depletion and thus cause adverse impacts on aquatic organisms.

2.1.71 In allowing excess waste water to be discharged to local watercourses, CSOs avoid, firstly, waste water 'backing up' and flooding streets and properties (with corresponding health risks) and, secondly, the inundation of treatment plants and the consequent disruption of treatment processes. This can cause more environmental damage than the CSO discharges themselves (Defra, 2012).

2.1.72 Storm sewage discharges from CSOs may contain significant loads of a wide variety of pollutants, including bacteria and viruses, oxygen demanding and toxic pollutants, as well as persistent materials such as heavy metals and Polycyclic Aromatic Hydrocarbons (PAHs). The presence of gross solids of obvious sewage origin is also a frequent problem. Although only discharged over short periods of time on an infrequent basis, these pollutants can seriously compromise many beneficial uses of receiving waters such as fisheries, shellfisheries, bathing and recreational water use, as well as the perceived amenity value of waters (CIWEM 2004).

2.1.73 CSOs have been linked to adverse, short-term (<5 day) impacts on water quality. A study conducted by the City of London Port Health and Environmental Services Committee (Lane et al. 2007) to investigate the risk of ill health amongst recreational users of the Thames found evidence that discharges of raw or combined sewage and rainfall runoff directly affected the microbiological water quality of the tideway resulting in peak levels of pollution concurrent with and following CSO discharges. However, the results also suggested that the impacts on water quality are relatively short lived and that it takes between two and four days for levels of indicator organisms to return to background concentrations following CSO discharges (Figure 2-7). A study of an exceptional CSO discharge event in the Seine (following heavy rainfall) found that the downstream concentrations of Faecal Indicator Bacteria (FIB) were more than two orders of magnitude higher than during normal (dry weather) conditions; however, concentrations in this study reduced more quickly, and dropped by 66-79% within 13 hours (Passerat et al. 2011).

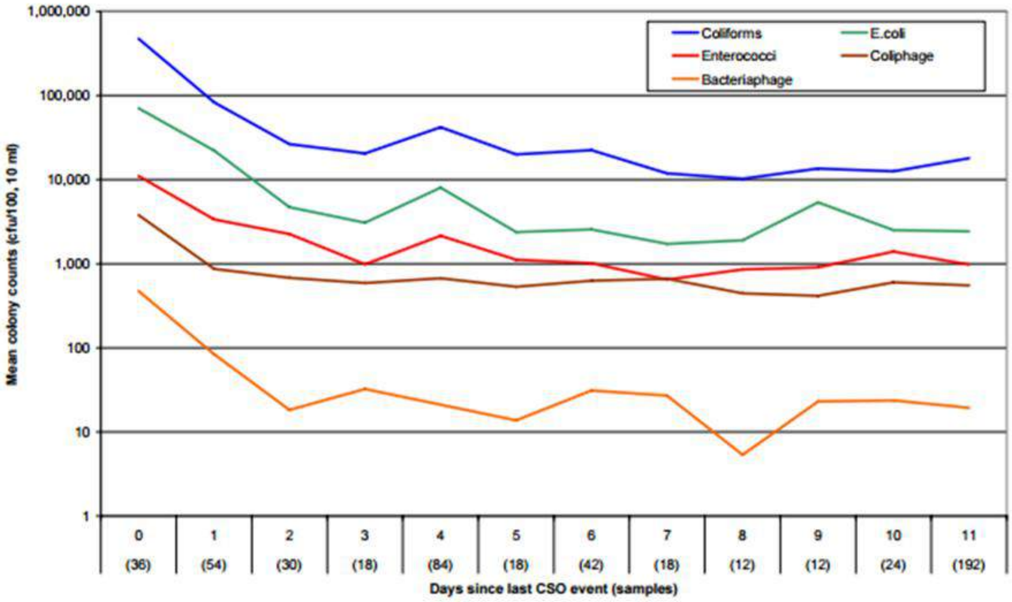


Figure 2-7 Microbiological water quality in the Thames relative to time since the last local CSO discharge

(Source: Lane et al. 2007)

2.1.74 Establishing concentrations of pathogens following a CSO discharge to the sea is considerably more difficult than it is to rivers, due to the relative complexity of the hydrodynamic environment. A study modelling CSO discharges to the sea found that the impact of discharges on levels of *E. coli* in the sea was highly dependent on local hydrodynamics (De Marchis et al. 2013). This study also found that a typical discharge concentration (of 10^4 MPN, or 'Most Probable Number' of viable cells) of *E. coli* caused the European limit for this pathogen to be exceeded along 50% of the modelled shoreline (of length ~ 4km) in the immediate aftermath of the discharge event.

2.1.75 There is also evidence to suggest a relationship between recreational water exposure and waterborne illness which may result from the ingestion of polluted water, immersion in recreational water, and contact through skin or inhalation. However, it is difficult to determine the infectious dose of a particular pathogen and the risk of infection is related both to exposure to organisms in sufficient numbers to cause infection as well as the susceptibility of those exposed. In the Thames study (Lane et al. 2007), 77% of those reporting an illness had rowed on the river in the three days following a CSO discharge. However, the small numbers of cases reported and the relatively high frequency of discharge events combined to reduce the statistical significance of these results by increasing the potential likelihood of illness events occurring by chance. Moreover, as Lane et al. (2007) pointed out, the population under study is not the same as the general population in that they are predominantly fitter and healthier as a result of their recreational activity. In addition, it is possible that continual exposure to what may be considered polluted water on a high frequency basis may raise the threshold of immune response (the point at which the immune system elicits an immune response resulting in symptoms) within this population, thus increasing the potential for asymptomatic infection. Since the study addressed self-reported illness based on symptomatic illness, these sub-clinical infections would not be detected.

2.1.76 A study of a CSO discharge to the sea following an extreme (1 in 20 year) rainfall event in Copenhagen reported an observed disease incidence rate of 55% amongst swimmers (Ironman competitors, n=1,312) exposed to high levels of five pathogens (Andersen et al. 2013). In this study, both a 3D hydrodynamic model and a drainage model were applied to estimate dilution of (and thus exposure) to the pathogens concerned. These models, and the large number of cases involved, enabled a more detailed, quantitative microbial risk assessment to be undertaken. This technique is the usual means of assessing the health impacts from exposures to a pathogen, and would not normally have been feasible without water quality data collected at the time of the discharge.

2.1.77 There is also a small but growing body of research on the wider ecological impacts of CSO discharges. In extreme cases, discharges can result in fish mortalities, the destruction of other aquatic life, and render shellfish unfit for human consumption (US EPA 2011). The impact of CSO discharges tends to be most pronounced on communities close to the stream bed (hyporheic system), where nutrients can accumulate and remain over the long-term (Hynes 1983). Downstream of CSOs, pollution tolerant species become common but overall species richness is reduced (Lafont et al. 2006).

Trends

Present day trend: ─ (low confidence).

Future trend: ↑ (low confidence).

2.1.78 Because the routine monitoring of CSOs in the UK has only just begun (outside of ‘sensitive areas’ under the EU WWTD), there is a high level of uncertainty over the present scale of the problem and the magnitude of future trends. Based on information from water companies and the Environment Agency, Surfers Against Sewage (2014) estimated that there were 1,500 separate pollution incidents from 786 separate sewer overflow discharges in 2014 (England and Wales, May to October bathing season). There is also uncertainty over the number of CSOs in the UK: the total estimate for England and Wales is 25,000 (Thompson 2012), and 6,000 for Scotland and Northern Ireland, giving a total of approximately 31,000 (MCS 2008). NGOs estimate that roughly a quarter of these are monitored (MCS 2008), with effort focused on existing monitoring requirements (such as in shellfish waters, or increasingly, in bathing waters).

2.1.79 Two key drivers of CSO discharges are population growth and urbanisation, both of which are expected to continue this century (ONS 2015). As cities, towns, and villages grow and new developments are established, the demand for sewage systems increases. Furthermore, urbanisation has increased the flashiness of runoff to sewerage systems, and thus reduced their capacity for carrying sewage (HM Government 2012). Continuing patterns of urbanisation also have the potential to interact with some of the negative impacts of climate change (Semadeni-Davies et al. 2008; See also ‘Impacts of climate change’ below).

Impacts of climate change

2.1.80 Climate change is already a major pressure on waste water infrastructure, and it is predicted that wetter winters, more intense rainfall events, and greater climate variability in general could lead to greater frequency of overflows into water courses (Defra 2010, Figure 2-7). There is some evidence that both the duration and frequency of CSO events will increase, but that the sensitivity of these systems to climate change depends on their ability to store excess water (Bendel et al. 2013, Fortier and Mailhot 2015).

Table 2-6 Overview of climate change impacts on Combined Sewer Overflows

Direction of impact	Climate variable		Mechanism(s)
	Present trend (confidence)	Future trend (confidence)	
Exacerbating	Winter precipitation		Increased discharge and runoff to sewage systems
	↑ (Low)	↑ (Low)	
	Summer precipitation		Reduced flushing and dilution
	↓ (Low)	↓ (Medium)	
	Rainfall intensity		Flashier, more frequent runoff
	↑ (Low)	↑ (Medium)	
	Temperature		Reduced assimilative capacity of environment
	↑ (High)	↑ (High)	

2.1.81 Studies suggest that winter rainfall has become more intense in the UK (Jenkins et al. 2008, Jones et al. 2013). Although projections for future changes in rainfall intensity are highly uncertain, recent research (Kendon et al. 2014) suggested that the intensity of winter rainfall could increase (by 2100, RCP 8.5 scenario). Should more intense rainfall events occur at this time of year and in areas where water tables are high and some soils are saturated, they would likely result in increased surface water runoff to sewage systems. The potential need for CSO discharges would thus increase.

2.1.82 Where it is available, CSO monitoring data suggests that annual volumes of CSO discharges are extremely sensitive to extreme rainfall events (have high annual variability in general). For example, a 1 in 25 year, 24 hour rainfall event in May 2014 resulted in a 41% increase in total annual volume for CSO discharges in the Michigan area (versus the previous year, US EPA 2007). In the same data, the measured frequency of CSO events does not correlate well with the total annual volume of water discharged (US EPA 2007), emphasising: a) that there is no 'typical' CSO event, and b) that real-time monitoring of not just the frequency of CSO events, but their volume and/or duration, is critical to quantifying their impact.

2.1.83 Surface water runoff can make a substantial contribution to CSO discharge volumes; for example, a study of the most intense rainfall event of the year in Paris in 2008 found that surface water runoff contributed to 85-92% of the total CSO discharge volume (Passerat et al. 2011). Continued population growth and urbanisation in the UK is expected to increase the flashiness of this surface water runoff (due to increased paving, hardstanding etc.), and therefore lead to more frequent CSO discharges in these areas (should we fail to routinely fail to incorporate Sustainable Urban Drainage Systems, SUDS or other similar outcome measures in new developments, see 'Adaptation'). CSO discharges would also increase if the capacity of existing wastewater treatment is not expanded to accommodate increases in the volume of sewage arising from population growth in the service area (MCS 2008).

2.1.84 Climate change will also reduce the capacity of the water environment to assimilate incoming sewage from CSOs, thereby magnifying their impact. Low flows (Q95) levels are projected to decrease by 20-40% (A1B medium emissions scenario, 2050s, Prudhomme et al. 2012), which will reduce the flushing and dilution of pollutants. This will likely result in increased nutrient uptake over a shorter length of river, and therefore result in a more localised enrichment at CSOs (Meyer et al. 1999). However, increases in temperature will also increase productivity, and nutrient cycling (Meyer et al. 1999). Thus there is uncertainty over the extent to which reduced annual or seasonal river flows may also require higher standards of sewage treatment in order to meet statutory environmental requirements (HM Government 2012). This uncertainty will be more apparent where a greater proportion of the nutrient load comes from CSO discharges, rather than (more predictable) continuous releases from wastewater treatment works.

2.1.85 No studies have yet modelled future CSO discharges under climate change and related these discharges to impacts on human health. However, the European Centre for Disease Prevention and Control (2011) offer an online tool for quantitative microbial risk assessments of climate change impacts on waterborne disease. This tool allows users to test the impact of various simple changes to climate (e.g. +2°C temperature rise, +20% precipitation increase) on the concentration of observed pathogens at a particular discharge location.

Adaptation

2.1.86 As the efficacy of sewage treatment improves, the proportion of total pollutants accounted for by CSO discharges into watercourses is increasing (Weyrauch et al. 2010). Performance under the strengthened EU Directives (particularly those pertaining to coastal areas) will therefore increasingly depend on the ability of Member States to reduce and/or eliminate CSOs. This can be achieved via a combination of separate sewage systems for rainwater and wastewater, surface water runoff control, real-time monitoring data, and infrastructure improvements where required.

2.1.87 The EU WWTD requires that pollution from CSOs is limited, and that unsatisfactory intermittent discharges are improved. ‘Unsatisfactory’ includes CSOs that operate in dry weather conditions, pollute groundwater, or cause adverse physical, chemical, visual or aesthetic impacts. A CSO can also be deemed ‘unsatisfactory’ if it causes a breach of relevant EU Directives. The WWTD (and preceding Directives) has driven a widespread improvement in water quality in coastal areas, where previously the discharge of raw sewage into the sea was commonplace. The WWTD has also driven improvement in, or closure of, a large number of CSOs (e.g. 4,700 brought up to standard in England and Wales between 2000 and 2005, Defra 2002). This can include the installation of mesh screens to prevent sewage related debris from being discharged, or increasing the diameter of the downstream sewer pipe. The improvements form part of an ambitious, long term investment programme that is reviewed every five years, which could take another 20-25 years to complete (Scottish Water 2007).

2.1.88 Measures to reduce CSO discharges can also target surface water runoff. Separate sewage systems for rainwater and wastewater have been constructed in new developments in the UK for over 50 years. Unlike a combined system that handles both types of water, they divert rainwater from roofs, roads and driveways into a separate network of pipes that discharges to the nearest river or stream. This alleviates pressure on CSOs at treatment works by reducing the volume of water entering the wastewater system during intense rainfall.

2.1.89 Sustainable Urban Drainage Systems (SUDS) take this further by ‘naturalising’ catchments through the use of porous paving surfaces, and by diverting the runoff away from direct entry into water bodies or stormwater drains. The volume and rapidity of runoff during precipitation events (which can be a particular problem in urban areas, Whitehead et al. 2009) is further reduced, lowering the risk of CSO discharges, and flooding. Different types of SUDS attenuate runoff in different ways. Most types of SUDS have good potential to reduce the absolute volume of runoff, and good potential for reducing the rate of runoff during more common rainfall events (such as 1 in 2 year events); yet many perform less well during less common rainfall events (such as 1 in 100 year events, CREW 2012).

2.1.90 Due to the cost of retrofitting SUDS to existing systems, SUDS are likely to be a more suitable option for surface water management in new developments. To encourage their wider deployment, the UK Government has recently confirmed that SUDS are now “the expectation” for new developments of 10 homes or more under national planning policy in England (as of April

2015, HM Government 2014). A similar requirement has existed in Scotland since 2011. In Northern Ireland, a strategy for SUDS was developed in 2011, while the Welsh Government consulted on its proposed approach to SUDS in 2015.

2.1.91 However, SUDS are unlikely to have the capacity to mitigate the full impact of CSO discharges. Commenting on the role of adaptation options for CSOs, Defra (2010) noted that "[Surface water management] will not always be able to prevent impacts of intermittent discharges from CSOs and new infrastructure projects will be needed to address them". This echoed the findings of UKWIR (2003), who specifically highlighted CSOs as one of two factors likely to drive a significant reduction in sewage systems performance under future climate change (the other factor highlighted was flooding).

2.1.92 The recent introduction of CSO monitoring is an important first step in an adaptation approach to this issue, as it will allow associations between CSO discharges and climatic events (and eventually climate change) to be established for the first time. Although some studies have examined these associations overseas, the sensitivity of CSO impacts to local hydrology (De Marchis et al. 2013) illustrates the importance of undertaking UK-specific research in this area.

2.1.93 Any reduction in domestic water use could also reduce runoff volumes to sewage works. Defra (2010) estimated that savings from water efficiency education, stringent building codes for new homes, refurbishment of existing housing and promotion of water efficient devices and appliances could amount to a reduction in required sewer and treatment capacity for England of greater than 1 billion litres per day; however, it is also later concluded that "[demand management] …will not be sufficient to significantly reduce future demand for waste water treatment capacity" (Defra 2010).

2.1.94 Ultraviolet treatment at wastewater treatment works is also being increasingly used to reduce the impact of CSOs (see Section 2.1.81).

Valuation case study #2: CSOs and impacts on human health – the case of the Thames Tideway Tunnel

2.1.95 The link between CSOs and health costs is set out in the impact pathway in Figure 2-8. However, as discussed in the sections above, it is not known with any certainty how the frequency of CSO overflows are likely to change in future and it is difficult to quantify the impact of climate change on future overflow rates. As such, systematic monitoring of CSO overflows would be required before the potential impacts of climate change could be predicted.

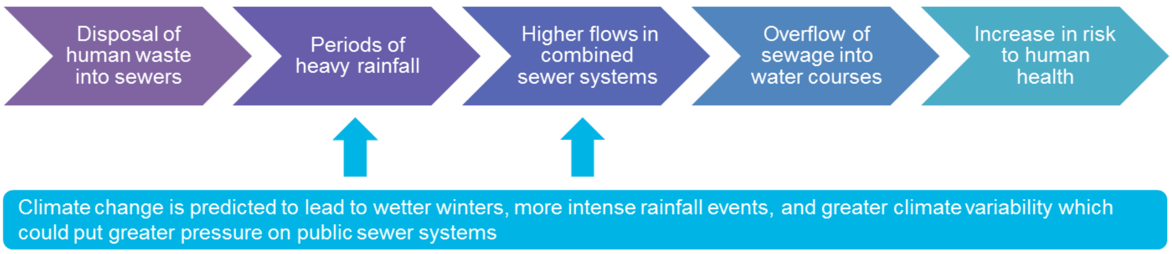


Figure 2-8 Links between combined sewer systems and human health costs

2.1.96 Much of the available analysis on the costs of CSO discharges relates to research undertaken as part of the appraisal of the Thames Tideway Tunnel. The Thames Tideway Tunnel is a proposed 25 km (16 mile) tunnel running mostly under the River Thames through central London, intended to provide storage and conveyance of combined raw sewage and rainwater discharges that currently overflow into the river. This project will tackle the problem of overflows from the capital's Victorian sewers for at least the next 100 years, and enable the UK to meet European environmental standards (Thames Tideway Tunnel, 2015).

2.1.97 The Thames Tideway Strategic Study (TTSS) was set up in 2002 as a three year project to assess the wider environmental impact of intermittent discharges of CSOs into the Thames Tideway. Discharges from London CSOs contain a mixture of sewage and runoff and under conditions of heavy rainfall the discharges can also contain large quantities of re-suspended sediment, litter and grease. Over time the frequency of operation of the CSOs and the quantity of storm sewage discharged has increased to the point where, on average, discharges from the catchment occur as often as 60 times per year and are frequently caused by moderate rainfall (Thames Water 2012).

2.1.98 The Thames Tideway Tunnel has been devised in order to intercept the key CSO discharges and convey them downstream for treatment prior to discharge. A study of the potential tunnel identified three key issues arising from CSO overflows in the Thames (Thames Water 2012):

1. **Human health risks** by increasing concentrations of pathogenic bacteria which potentially pose risks to users of the river. In order to quantify the potential health risks, WHO standards were used to define a bacteriological threshold above which water users were exposed to 'possible health risk'. Currently, some 20 million cubic metres of storm sewage are discharged annually from all the CSOs, with some individual discharges in excess of a hundred thousand cubic metres. As a result, the threshold is breached following CSO discharges giving rise to approximately 120 days of possible health risk per year.

2. **Biodiversity loss** by reducing Dissolved Oxygen (DO) levels in the river potentially resulting in the death of adult fish and fish fry (also see Section 2.1.26). A specially commissioned study of fish responses to low DO exposure reinforced some empirical standards based on existing water quality data. These were used in association with water quality models to estimate an allowable pollution load and thus support sustainable fish populations.

3. **Reductions in the attractiveness of the environment** due to quantities of solid material being discharged into the Thames and deposited on the foreshore. An estimated 10,000 tonnes of screenable sewage derived solids is discharged through CSOs to the Tideway each year. The assessment of the impact of the CSO discharges showed that 35 caused significant aesthetic pollution and guidance states that unsatisfactory CSOs should receive screening prior to discharge.

2.1.99 The Thames Tideway Tunnel project aims to ameliorate the negative impacts of CSOs on human health, wildlife and the aesthetic value of the river. In detail, the project aims are: a reduction in the quantities of pathogenic organisms in order to reduce the potential health risk to recreational water users; the prevention of major falls in DO levels of benefit to all biota, but particularly to fish populations including through reduced fish kill incidents and adverse impacts on fish behaviour, and thereby enhancing the sustainability of fish populations; and a substantial reduction in the amount of sewage-derived litter deposited on the foreshore and in the river - so avoiding potential health risks and aesthetic nuisance, and supporting public enjoyment of the river.

2.1.100 In 2005, the TTTS Cost-Benefit Working Group published a report on the costs and benefits associated with various options for achieving water quality and ecology objectives for the Tideway (Thames Water, 2005). The costs and benefits were assessed by means of three main studies:

- A stated preference survey of 1,214 Thames Water customers with the objective of identifying and valuing individuals' willingness to pay (WTP) for the non-market benefits resulting from the implementation of the Tideway solution options.

- An environmental costs study to evaluate the non-market environmental costs attributable to each of the Tideway solution options.

- A market evaluation study to identify the potential market benefits arising from the Tideway Solutions identified.

2.1.101 The stated preference study (Thames Water, 2005) found that households' were willing to pay:

- £1.80 per year for reducing the amount of sewage litter in the Thames by one percentage point;
- £0.40 per year for reducing the number of elevated health risk days by one; and
- £1.50 per year for each potential fish kill avoided.

2.1.102 WTP to avoid a day of elevated health risk due to CSO discharges into the Thames was £0.40 (Thames Water 2005). The WTP survey was undertaken in 2003 and was subsequently updated in 2006 to take into account additional details in the scheme design. However, the revised figures are not readily available.

2.1.103 Also as part of the Thames Tideway Tunnel project, NERA undertook an analysis of the health impacts of CSO discharges in the Thames using Quality Adjusted Life Years (QALYs) to quantify the economic costs. Although we did not have access to the original study, Binnie (2012) reported that NERA analysed the values of the potential health benefits based on the number of recreational users, annual risk of infection, average duration of illness (expressed as % of a year), and value of a QALY, leading to an estimated annual cost of £22,000. Binnie (2012) also estimated that this is around 60% of the cost estimates derived using the WTP approach.

2.1.104 A cost benefit of the project undertaken by Defra in 2011 used these WTP estimates to consider the potential returns on the project. According to their calculations, the total present value benefits were estimated to range from £2,969m to £5,058m, although it was considered that the actual value was likely to lie towards the higher end. This is relative to estimated costs over the whole life of the Thames Tunnel Project of £4,061m, giving a potential benefit:cost ratio within the range of 0.73:1 to 1:1.25 (Defra 2011).

2.1.105 The Thames Tideway case study therefore suggests that there are likely to be significant co-benefits to addressing CSO discharges, in terms of human health, wildlife, and aesthetic value. This specific case suggests that, together, these benefits may outweigh the costs of achieving them.

2.1.106 While the valuation estimates undertaken as part of this study are useful for illustrating the potential costs of CSO discharges on human health, it is important to note that the Thames may not be representative of all UK rivers and it is difficult to draw any firm conclusions at a national level. As such, there is insufficient evidence from which to develop a robust value transfer function which could be used to estimate the costs to human health of CSO discharge events more widely.

Dissolved Organic Carbon (DOC)

Description of the problem

2.1.107 The decomposition of peatland habitats leads to the release of organic carbon from soils. Dissolved Organic Carbon (DOC) concentrations affect water colouration, the toxicity of certain metals and result in changes in ecological conditions.

2.1.108 DOC is defined as the organic carbon in water that can pass through a filter (commonly 0.45 micrometres). Inorganic forms of carbon that can pass through this filter, including bicarbonate, carbonate and carbon dioxide, are termed Dissolved Inorganic Carbon (DIC). Forms of carbon that cannot pass through the filter are known as Particulate Organic Carbon (POC). DOC in both terrestrial and marine systems is a hugely important component of the global carbon cycle, although here we mostly deal with DOC in terrestrial freshwater systems, and focus on upland peats.

2.1.109 Organic carbon is released by the aerobic decomposition of organic matter in the surface layers of the soil (Bateman and Georgiou 2006). Within the UK, DOC typically represents the main component of this release (Grand-Clement et al. 2013). For example, in Welsh unblocked peatlands, the ratio of DOC to POC ranges between 15:1 and 50:1 in conditions of low and high rainfall respectively (Wilson et al. 2011). POC can, however, dominate in heavily eroded peatlands and the two compounds are closely interlinked, with POC reacting in water and transforming into DOC. Recent work on headwater systems with high POC concentrations has found that POC:DOC ratios decrease rapidly downstream which may be due to degradation of POC to DOC in stream systems (Evans et al. 2013). For clarity, the overview in this section focuses on DOC although it is noted that POC is an important and interrelated issue.

2.1.110 DOC results in colouration of the raw water supply (raw water being natural water found in the environment). The colour in the water arises from humic substances of high molecular weight absorbing visible light more strongly at the blue end of the spectrum, giving the water a brown colour (Evans et al. 2005). Note that 'discolouration' at the tap is often a separate issue, and caused by the corrosion of mains or service pipes. Water companies with upland catchments have recorded an increase in Hazen Units of the raw (inlet) water at their treatment works, with higher levels of Hazen costing more to treat (at its worst, colouration can render the water untreatable). Some water companies have also reported spikes in Hazen following extreme weather conditions (see 'Impacts of climate change' below). However, the exact relationship between colour and DOC varies between peat layers, land management regimes, and even with time, making direct attribution of specific Hazen levels to DOC concentrations difficult (Wallage and Holden 2010).

2.1.111 Higher DOC levels could modify the toxicity of certain metals in the water environment. This is because DOC facilitates the transportation of metal contaminants from soils in soluble organic complexes, particularly iron and aluminium (Evans et al. 2005 and Lawrence et al. 2013, respectively). Thus there may be indirect impacts of DOC on fish, via reduced exposure to toxic metals (Raymond and Bauer 2001, Evans et al. 2005). However, the reservoirs of these metals in the environment will decrease over time as catchments recover from historic atmospheric deposition. Moreover, the affinity of DOC to binding with metal has been shown to vary based on the degree of anthropogenic influence at the sample collection point (Baken et al. 2011), suggesting that a simple rule of thumb over any ameliorative impact of DOC on toxic metals will be difficult to establish.

2.1.112 Higher DOC levels can also bring about changes in aquatic ecological conditions. These include a reduction in the penetration of light into the water column, which reduces the depth of macrophyte growth but protects aquatic invertebrates from ultra-violet radiation (Smith et al. 2011). Higher DOC levels can also act to shallow the thermocline, increase the temperature in the epilimnion (the upper layer of water in a lake), and reduce the temperature and DO levels in the hypolimnion (the bottom layer, Stasko et al. 2012).

Trends

Present day trend: ↑ (high confidence).

Future trend: ↑ (low confidence).

2.1.113 While there are limitations in monitoring of DOC in water bodies, available evidence suggests that DOC concentrations have risen. While various drivers of DOC release have been proposed, the increase is most likely attributable to decreased acid deposition.

2.1.114 DOC concentrations in water bodies are not part of the assessment criteria under the WFD and thus are not captured by the water quality datasets described earlier (Section 0). However, DOC concentrations are monitored indirectly under the WFD, to determine the bioavailability of certain metals (within the Metal Bioavailability Assessment Tool M-BAT, UKTAG 2014a). Monitoring is also used to distinguish 'clear' catchments from 'humic' catchments in determining the relevant WFD standards for pH (NIEA 2014d). There are also a number of specific upland sites at which DOC is regularly monitored, which form part of a UK-wide peatland network (Evans et al. 2011) that is moving towards a complete spectrum approach, logging all relevant water parameters in-stream (Holden 2015). While spatial coverage is uneven, SEPA has maintained a high-density network of sites in the north east of Scotland since the 1980s. Monitoring tends to be focused on soil types and in landscapes where DOC concentrations and fluxes are at their highest, although a recent study (Parry et al. 2015) focused on undisturbed catchments.

2.1.115 Pooling the data from these sites and those monitored by others (FRS-Freshwater Laboratory, CEH, Yorkshire Water, UK AWMN, UK ECN, Forestry Commission), Worrall et al. (2004) discerned a significant upward trend in DOC at three-quarters of sites (n=198) of 0.17 mg C/litre/year. This linear, monotonic trend has been re-confirmed by a recent update of the trends in hydrochemistry monitoring data (Monteith et al. 2014), and has been observed in many industrialised countries worldwide. Rates of increase are greatest in the most organic-rich catchments (Freeman et al. 2001).

2.1.116 Various drivers of the long-term trend in increased DOC have been proposed, including: declining acid (sulphur) deposition, increasing nitrogen deposition, land use change, land management changes, and climate change (via an enzymic latch mechanism). The rate of observed DOC increase at monitoring sites has been proportional to the levels of organic carbon stored at those sites (Evans et al. 2005). This implies the driver of the increase is spatially consistent and acting uniformly across the UK. Observational studies in the field, and in laboratories, have converged on the hypothesis that a reduction in acid deposition (acid rain, but also dry deposition, Monteith et al. 2007) is behind the widespread, long-term rise in DOC levels. The implementation of emissions controls has raised the pH of soils towards more neutral conditions, in which soil organic carbon is more soluble and thus more vulnerable to transport. This does not preclude the influence of other drivers of DOC levels at the site scale, and many authors on the subject stress the importance of these drivers on short term increases in DOC in particular landscape contexts (e.g. Futter et al. 2011). For example, a recent study of undisturbed catchments in the Pennines (Parry et al. 2015) suggested that mean slope was the most important determinant of DOC export, with vegetation type found to be less important.

2.1.117 Land management is also an important control of stream water DOC concentrations and is therefore a potential influence on water colour (Yallop et al. 2010). Other factors include proximity of the peatland to a stream (e.g. Bishop et al. 1994) and other hydrological processes (Dawson et al. 2008). Additional drivers acting at local scales can include peatland drainage and overgrazing, while over large areas, increases in atmospheric CO_2, occurrence of severe droughts, and eutrophication can all affect DOC levels (Grand-Clement et al. 2013; Smith et al. 2011). Previous difficulties in documenting land management in the uplands meant that evidence recording the risk from various land management techniques was conflicting, or the evidence showed no management impact on DOC (Evans et al. 2005). At sites where DOC levels are simply recovering to pre-industrial conditions in the absence of harmful management practices, there may be relatively little that changes in land management would achieve in terms of attenuating DOC export.

2.1.118 However, the recently completed EMBER project revealed that heathland burning, in which controlled burns are conducted to improve conditions for red grouse (*Lagopus lagopus scotica*), thereby managing the land for gun sports, lowers the water table and thus could possibly interact with climate change to exacerbate carbon fluxes to water bodies (Brown et al. 2014). Burning also increased the local temperature of the soil (the 'microclimate') which could further encourage carbon release. Natural England's Upland Evidence Review on the burning of blanket bog also found "strong evidence" that burning results in increased water colouration and/or DOC in peatland watercourses (Glaves et al. 2013). These are important findings as, prior to this study, the evidence for an impact of burning on DOC was equivocal (Holden et al. 2012). Evidence on the efficacy of management addressing peatland degradation (e.g. ditch blocking) has also recently been collated (Parry et al. 2014) but, due to the relative infancy of these techniques, can only describe short-term impacts (see ‘Adaptation’ below).

Impacts of climate change

2.1.119 The increase in average temperature as a result of anthropogenic climate change has been ruled out as the main driver of the recent, long-term increase in freshwater DOC levels, but higher temperatures are likely to increase production (and thus release of DOC from soils).

2.1.120 While recent trends in DOC concentrations are not attributable to climate change (Monteith et al. 2007), in the long term, acid deposition appears to be stabilising, and the main determinants of DOC levels in future may increasingly become climate and land management (Smith et al. 2014). Higher temperatures are likely to increase the production (and thus release) of DOC from soils, and this impact has already been demonstrated at the laboratory, field and catchment scales (Evans et al. 2005). For upland landscapes with substantial coverings of frost and/or snow in winter, higher temperatures will increase soil exposure to ultraviolet photodegradation (Stasko et al. 2012). Finally, any increase in the frequency of fires will leave soil carbon vulnerable to erosion and thus runoff losses of DOC. However, the most substantial impact of climate change is likely to be through changes in the frequency of short-duration drought events and changes in the intensity of rainfall. Drought drives the production of DOC by drawing down the water table and leaving peaty soils vulnerable to intense rainfall events, which mobilise the organic carbon as DOC and thus also cause colouration events (Worrall et al. 2004). Increased colouration has been observed in the field (Curtis et al. 2014), computer modelling experiments (Li 2014) and in mesocosm experiments (involving direct manipulation of the water table itself, Lou et al. 2014). Most notably, the severe drought of 2004-06 caused elevated DOC levels in nine AWMN (Acid Waters Monitoring Network) lakes, but these have shown signs of recovery (Curtis et al. 2014).

2.1.121 Climate change will bring both: a) an increased risk of summer drought events, and b) more intense rainfall (Murphy et al. 2009, Table 2-7), and the severity of these impacts will be a function of how quickly and completely upland peats can recover in the intervening periods between these events. In computer modelling experiments, rates of peat erosion were found to increase with projected rises in temperature, although they were more sensitive to the local land management regime (Li 2014).

Table 2-7 Overview of climate change impacts on dissolved organic carbon levels

<table>
<tr><th rowspan="2">Direction of impact</th><th colspan="2">Climate variable</th><th rowspan="2">Mechanism(s)</th></tr>
<tr><th>Present trend (confidence)</th><th>Future trend (confidence)</th></tr>
<tr><td rowspan="6">Exacerbating</td><td colspan="2">Drought</td><td rowspan="2">Water table draw down, leaving soil carbon more vulnerable to being mobilised</td></tr>
<tr><td>↑ (Low)</td><td>↑ (Low)</td></tr>
<tr><td colspan="2">Storminess</td><td rowspan="2">Increased dissolution (and suspension) of soil carbon and transport into streams and rivers</td></tr>
<tr><td>- (High)</td><td>↑ (Medium)</td></tr>
<tr><td colspan="2">Temperature</td><td rowspan="2">Increased soil production of DOC; reduced frost days and snow would increase exposure to ultraviolet photodegradation, leading to higher runoff; increased frequency of fires would also leave soil carbon vulnerable to transport via runoff</td></tr>
<tr><td>↑ (High)</td><td>↑ (High)</td></tr>
<tr><td rowspan="2">Mitigating</td><td colspan="2">Winter precipitation</td><td rowspan="2">More winter precipitation (if not intense) would increase wetness and decrease vulnerability of soil carbon to mobilisation</td></tr>
<tr><td>↑ (Low)</td><td>↑ (Low)</td></tr>
</table>

2.1.122 Given the complexity of the mechanisms driving DOC levels in upland waters (Limpens et al. 2008), research into which landscapes or habitats might be more vulnerable to DOC export in the future is ongoing. DOC concentrations respond negatively to site elevation in all soil types, but more so in wetland peats than in organo-mineral types. This could be because wetland peats are more vulnerable to water table draw down, especially at lower elevations where precipitation is lower and evapotranspiration is higher (Monteith et al. 2015). It could also be because the increased wetness impedes respiratory losses. Successful identification of more resistant catchments would allow water companies to devise strategies for sourcing raw water that avoid those catchments worst affected by DOC export (Parry et al. 2015).

Adaptation

2.1.123 As much as 70% of peatland in England is physically degraded (Natural England 2010). The extent of degradation (and drying out) in landscapes like Exmoor is such that the local water company (South West Water) often has to resort to the use of pump-storage reservoirs during low flows (EUPN 2012). Physical features resulting from historic peat cutting and drainage include grips/drains/ditches, erosion gullies, bare peat (pans), underground peat pipes, and haggs.

2.1.124 Various restoration projects have sought to improve the condition of the peats by targeting these features, although those projects that have monitored DOC thereafter have tended to focus on ditch blocking (Parry et al. 2014). On the whole, these studies conclude that blocking ditches leads to reduced DOC concentrations in the ditches, and a national survey cited a reduction of 28% (Armstrong et al. 2010). This figure is certainly not consistent across the country, however, and DOC concentrations at the most intensively monitored site in the UK (Upper Wharfedale in the Yorkshire Dales) show no difference between blocked and unblocked treatments. The carbon may also be leaving the peat via an alternative (unmonitored) hydrological route. Because even the more established restoration projects have only recently been completed, all these studies essentially represent the short-term impacts of restoration, and they will therefore need to be monitored over longer periods to capture longer-term impacts. The wider benefits of peat in good condition - for water quality, greenhouse gas fluxes to the atmosphere, or sequestration - mean that all UK countries are engaged in policies to restore damaged or degraded peatlands (e.g. ‘Peatlands for the Future’ in Wales).

2.1.125 As an aside, with respect to climate change mitigation, DOC export can have a notable influence on the status of a peatland as a carbon sink or a carbon source; for example, Billet et al. (2004) found that the inclusion of DOC (and other riverine) loss was sufficient to turn a catchment at Auchencorth Moss in Scotland from an apparent carbon sink to a substantial (83 g C/m2/yr) carbon source (ECOSSE 2007). Thus peatland restoration projects often monitor and manage for DOC levels explicitly. A recent Defra research project (Evans et al. 2013) examined the fluvial loss of all forms of carbon in peatlands (including both DOC and POC). The same authors determined that all forms exported from peatlands will eventually be emitted as carbon dioxide to the atmosphere, regardless of the downstream design of drainage networks, reservoirs or treatment works. They therefore concluded that fluvial carbon losses should be included in peatland greenhouse gas budgets, and that there is no substitute for reducing these losses at source, via improved management (using techniques like re-wetting).

2.1.126 The impact of re-wetting on the net flux of greenhouse gases is still an area of active research, and is possibly context dependent, although most studies report a net benefit in terms of climate change mitigation potential. Studies often report that pristine, undisturbed or restored peatlands are a sink for carbon and a source for methane, while drained peatlands become a source for carbon (Vanselow-Algan et al. 2015). A recent global meta-analysis concluded that rewetting resulted in a net flux of greenhouses gases (converted to carbon dioxide equivalent) into the soil (Bonn et al. 2014), with particular sequestration benefits available for the conversion of cropland to fen, and eroded blanket bog to intact blanket bog. However, re-wetting following periods of severe drought may in fact accelerate carbon loss to the atmosphere (Fenner and Freeman

2011), via a biogeochemical cascade, and should the return period for these drought events continue to decrease, this impact could have the potential to destabilise carbon stocks.

2.1.127 Given the increasing risk to water supplies sourced in peatland ecosystems, peatland restoration may also come to feature prominently in 'Payment for Ecosystem Services' (PES) Schemes. The substantial lead in time for any land management intervention will mean that the establishment of a fair and successful scheme for peatlands will depend on the extent to which policy direction is consistently focussed on the long term. For landowners, the concern will be that changes to the uplands are executed that are beneficial for water companies, but that payments are not sustained once the initial boost in water quality or flashiness in flow is over. For water companies, their concern is that landowner interest and activity may not be sustained until the end of the project term (EUPN 2012). The role of an intermediary (such as the IUCN, Defra, or others including the third sector) in setting and monitoring the terms of such agreements will therefore be vital. The pilot phase of a new (voluntary standard) UK Peatland Code is underway (at the time of writing, August 2015), which will seek to establish the framework in which PES could operate. An ongoing Defra research project (Dickie et al. 2015) recently concluded that peatland extent is inadequately represented by existing national maps of land use / land cover; hence, the development of such maps (e.g. the new peat map for Wales, Evans et al. 2015) should be prioritised if peatland (as an asset) is to be promoted for inclusion in PES schemes.

Valuation case study #3: DOC, water colouration and impacts on the public drinking water supply

2.1.128 This case study considers the impacts of water colouration on the public drinking water supply and the costs that this imposes on water companies and, ultimately, their customers. In particular, the case study focuses on water colouration caused by degradation of upland (predominantly peatland) habitats.

2.1.129 Peatland habitats are an important source of water for many catchments. In the UK, approximately 70% of all drinking water is derived from surface water that comes mainly from upland catchments, which are often peat dominated. They also play a role in storing and filtering water which can remove impurities resulting in high quality water that is much cheaper to treat for drinking (Bain et al., 2011).

2.1.130 High levels of DOC and POC export can lead to colouration of raw water which incurs additional treatment costs for water companies that abstract water for the public drinking water system. This treatment addresses the odour and taste issues for users (Bateman and Georgiou, 2006). However, DOC can also react adversely with the chlorination process at treatment works to release Trihalomethanes (THMs), a hazard to human health (World Health Organisation 2005, Smith et al. 2011). As such, higher DOC levels in raw water resources require careful management at the treatment stage, which generates higher treatment costs for water companies. The link between peatland habitats and water treatment costs is set out in the impact pathway in Figure 2-9.

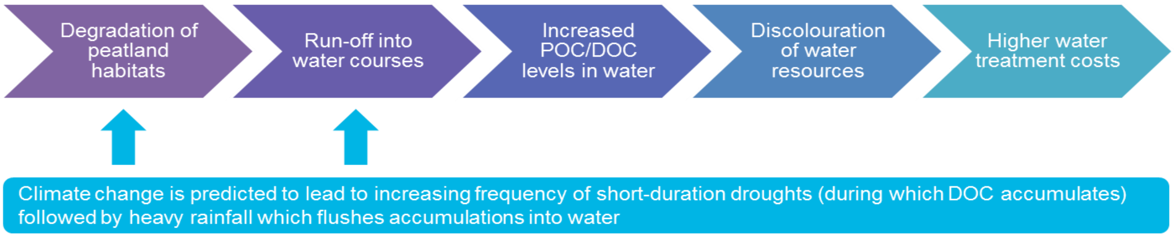


Figure 2-9 Impact pathway for peatland degradation and water treatment costs

2.1.131 In response to the rising trend in DOC, there is a growing awareness of the impact of DOC on the public water supply, particularly amongst water companies. UK water utilities are facing increasing costs associated with more sophisticated treatment process, such as coagulation, adsorption, and membrane filtration (Armstrong et al. 2010; Martin-Ortega et al. 2014). As a result of this, a number of catchments affected by colouration are designated as DrWPA Safeguard Zones, and a number of catchment level management programmes are being explored by water companies which aim to reduce raw water treatment costs through investment in peatland restoration projects (Morris and Holstead 2013). Ofwat has anticipated that "[n]early two-thirds of the money we expect the [water] companies to spend [on catchment management] over the period 2010-15 is for work that United Utilities, South West Water and Yorkshire Water will carry out to restore upland water catchments" (Ofwat 2011).

2.1.132 Quantifying the costs of a change in water quality due to a change in DOC concentrations is a conceptually straightforward exercise. Water colour is a major water treatment issue for water utility companies because the removal of colour imposes a direct financial cost. According to Whitehead *et al.* (2006), DOC removal represents the single largest cost to water utilities in the UK. As such, reduced colouration from peatland restoration has a direct financial benefit for the utilities in the form of reduced treatment costs (Martin-Ortega et al. 2014).

2.1.133 In practice, however, quantifying such costs can be a challenging exercise due to the difficulty in accessing cost data from water utility companies. In a review of the evidence and challenges associated with valuing water quality improvements from peatland restoration, Martin-Ortega et al. (2014) concluded that, "our review of the scientific and grey literature found no published data". Likewise, Bateman et al. (2011) concluded that "assessment of the avoided remediation costs of water purification which may come about by environmental improvement is complicated, as necessary information is typically considered as confidential by private water utilities".

2.1.134 It is nevertheless possible to estimate the avoided costs of measures (e.g. blocking drains to reduce peat wastage) to reduce colouration problems. However, these will vary on a catchment-to-catchment basis and are not known at a national level.

2.1.135 However, Tinch et al. (2010), drawing on estimates from an earlier study by Beharry-Borg et al. (2010) as well as information on the average daily costs of treating parameters such as DOC, showed potential benefits from avoided costs of treatment to be around £5 million over 10 years. In this example, the annual mean DOC concentration was 14 mg/L and projected to rise to 21 mg/L after 10 years under a 'do nothing' scenario. The study estimated that the associated increase in treatment costs per day would be approximately £146 for a typical treatment plant with a capacity of 60 ML/day; suggesting a value of around £0.35 per ML per 1 mg/L increase in DOC concentration.[2] The study then estimated the average cost of treatment per day, exclusive of any increase in DOC treatment costs, to be £1,316 and the total avoided costs of treatment for 10 years for a typical plant at £5.3 million.

2.1.136 In a similar vein, the IUCN Commission of Inquiry on Peatlands (Bain et al., 2011) reported that models developed by United Utilities and Yorkshire Water predicted an increase of one hazen per ML per day of water treated would result in an increase in treatment costs of between £0.10 and £0.20. They also added that the values are site specific and, after a threshold point, the costs of water treatment start to become increasingly significant; for example, modifications may need to be made to treatments works or new pipes may need to be laid. According to their research, a typical magnetic ion-exchange (MIEX) process added on to a conventional 10 ML three stage water treatment works may cost between £5 and £7 million to construct. In addition to the initial capital cost, MIEX is an energy intensive solution that greatly contributes to the operational cost and carbon footprint of the treatment process (Bain et al. 2011). Further details on some of the modelling behind these estimates can be found in Harlow et al. (2012).

2.1.137 A further study by Morris and Camino (2011) used an alternative approach to estimating the potential benefits of improving water quality through land management activities. Rather than looking at the potential benefits of reducing DOC levels through the associated reductions in

[2] This assumes that the total increase in cost per ML is equal to £2.43 (£146/60ML= £2.43ML) and the total increase in cost per ML due to a unit change in DOC concentrations of 1 mg/L is equal to £0.35 (£2.43/(21mg/L-14mg/L)).

water treatment costs, the authors looked at a range of recent studies which quantify the value of ecosystem services provided by wetland habitats (including peat bogs). The study was based on a review of recent meta-analyses of the value of ecosystem services provided by wetlands and aimed to estimate the monetary value of water quality improvements provided by wetland habitats. The study found that the marginal value of the improvement in water quality provided by an additional hectare of inland wetland habitat (this includes both 'bog' and 'fen, marsh, and swamp' habitats), was £292 each year. While not directly comparable to the other examples cited above because it considers a much wider range of benefits from wetland restoration than just changes in DOC levels, the findings provide an indication of the value of water quality improvements that may be achieved through wetland restoration projects in the UK.

2.1.138 A summary of the value estimates identified through the review is set out in Table 2-8.

Table 2-8 Summary of the results of the review of value estimates

Value estimates (unadjusted)	Aspect considered	Reference
£0.35 (£/ML)	Increase in treatment costs of a ML of water per day due to an increase in DOC concentrations of 1 mg/L	Tinch et al. (2010)
£0.10-0.20	Increase in treatment costs for an increase of one Hazen (water colour unit) per ML per day of water treated	Bain et al. (2011); Harlow et al. (2012)
£292 (£/ha/year)	Marginal value of water quality improvement provided by an additional ha of inland wetland each year	Morris and Camino (2011)

2.1.139 While several studies have attempted to quantify the value of changes in water quality as a result of upland land management, these estimates tend to be site-specific and have been generated using different methodologies. They are often based on average treatment costs in the absence of more detailed cost information from water companies. As such, it is recommended that a key area of further research could be to work with water utilities to develop a method for quantifying these costs and a value transfer function that could be used to more reliably estimate the benefits of land remediation interventions (in the form of cost savings to water companies) across different sites.

2.1.140 Due to the interconnectedness of ecosystems and their associated services, a change in the capacity of a habitat to provide a particular service can have significant knock on impacts on its capacity to provide other services. As such, there are potentially significant trade-offs and co-benefits arising from changes in water quality due to impacts on peatland habitats. In a chapter looking at the ecosystem services provided by mountain, moorland, and heath habitats, for example, the UK NEA provided a case study of the impacts on a range of services arising from peatland restoration on the Bleaklow plateau in the Peak District National Park (Van der Wal et al. 2011). Using this example, Table 2-9 sets out some of the potential trade-offs and co-benefits to improving upland water quality through restoration of peatland habitats.

Table 2-9 Potential trade-offs and co-benefits associated with improving water quality through upland peatland restoration

Ecosystem Service	Trade-offs (↓) or co-benefits (↑)	Explanation
Grazing sheep and cattle	↓	One of the key drivers of reductions in water quality in the uplands is overgrazing so peatland restoration schemes typically require a reduction in livestock densities.
Shooting grouse and game birds	↑ or ↓	Peatland restoration can potentially facilitate grouse shooting through increasing the total area of habitat

Ecosystem Service	Trade-offs (↓) or co-benefits (↑)	Explanation
		available although it can also require restrictions on moorland burning.
Flood risk	?	It is possible that re-wetting of peatlands could reduce the flashiness of downstream water runoff although quantitative evidence for this is limited at present.
Fire risk	↑	Enhanced soil moisture and vegetation cover through peatland restoration is likely to lead to reduced ignition risks and wildfire intensity.
Carbon storage and sequestration	↑	Degraded peatlands are a significant source of GHG emissions. Restoration can turn these habitats into substantial carbon sinks, although can also increase emissions of methane which are more damaging than carbon dioxide as a greenhouse gas.
Tourism and recreation	↑	Restoration is likely to improve the suitability for walking, and thereby the quality of visits, due to the stabilisation of peat surfaces.
Aesthetic landscape values	↑	Peatlands are an important landscape resource and greater plant cover on restored bogs combined with clearer water quality are likely to have increased visual appeal.
Biodiversity	↑	Enhanced vegetation cover on restored peatlands provides opportunities for a far greater number of species, both above and below ground.

2.1.141 As is clear from Table 2-9, there are a range of potential co-benefits (many of them nonmarket) associated with tackling water colouration problems arising from peatland degradation. In light of the potential benefits, a number of projects have been undertaken to restore peatland habitats in the UK and to measure the impacts on service provision (see Moors for the Future Partnership 2015 and the Peat Compendium 2015). While there are many projects which provide interesting case studies in terms of quantifying the co-benefits of tackling water quality issues in the uplands, there are relatively few studies which go as far as quantifying the economic costs and benefits.

2.1.142 One of the few examples that quantified the economic costs and benefits of such schemes is the Sustainable Catchment Management Programme (SCaMP), operated by United Utilities. The aim of this scheme is to protect and improve water quality by reducing the rate of increase in raw water colour by restoring peatland habitats within the catchments United Utilities operate in, thereby reducing future operational costs and the need for future capital investment in additional water treatment (United Utilities 2015). The first five years of the Programme (from 2005-2010) resulted in 98.6% of the 17,500 ha of SSSI being assessed as in favourable or unfavourable recovering condition, thereby generating a range of ecosystem service benefits. In the Peak District before SCaMP this had been assessed at 14% (United Utilities, 2015).

2.1.143 According to a recent cost-benefit analysis of the scheme, the total project costs were reported to be £7.4 million compared to benefits of £13.3 million as a result of improvements in water quality, biodiversity, recreation, and climate regulation (Hirst et al. 2012). Overall estimates of the benefit:cost ratio for the SCaMP project range from 0.55 under the most pessimistic scenario to 2.24 under the most optimistic scenario.

2.1.144 When reporting impacts on water quality, United Utilities cautioned that collecting and analysing water quality data is complicated by the fact that raw water quality from upland catchments is driven by the weather and seasonal variations. As such, it is normal to see significant colour increases during wet autumn periods and it can take several years before water quality benefits can be quantified with any degree of accuracy. Monitoring results at a sub-catchment level has indicated there is typically a statistical 'tipping point' two years after intervention as re-wetting dried peat initially releases more carbon in the form of colour before the natural biochemical processes are reasserted. At the present point in the SCaMP scheme, the results for several sub-catchments are indicating a slight decrease in colouration over time (see Figure 2-10, United Utilities 2014).

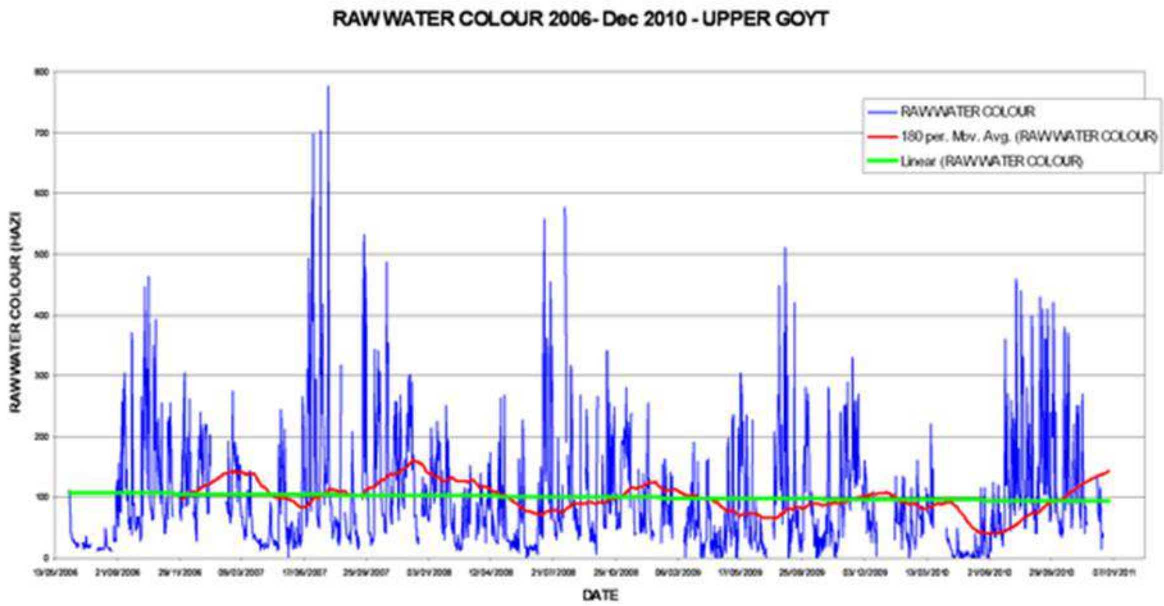


Figure 2-10 Raw water colour 2006-10 at SCaMP subcatchments

(Source: United Utilities 2014)

2.1.145 Looking ahead, United Utilities anticipates this trend to continue, and projects a slowing of the increase in colouration observed at treatment works, and possibly a drop in this colouration by 2020 (under the most optimistic scenario). United Utilities estimates that lower levels of colouration will lead to a reduction in ongoing annual treatment costs over time, as well as a reduction in capital expenditure due to a reduced need for additional water treatment measures (United Utilities 2014). If the increasing trend in colouration were to be stabilised, this would correspond to a saving of approximately £1.6m.

2.1.146 In addition to improvements in water quality, United Utilities has also reported that there are likely to be significant benefits in terms of carbon storage, biodiversity improvements, and recreation values arising from SCaMP which could potentially be significantly greater than the water quality benefits (Figure 2-11).

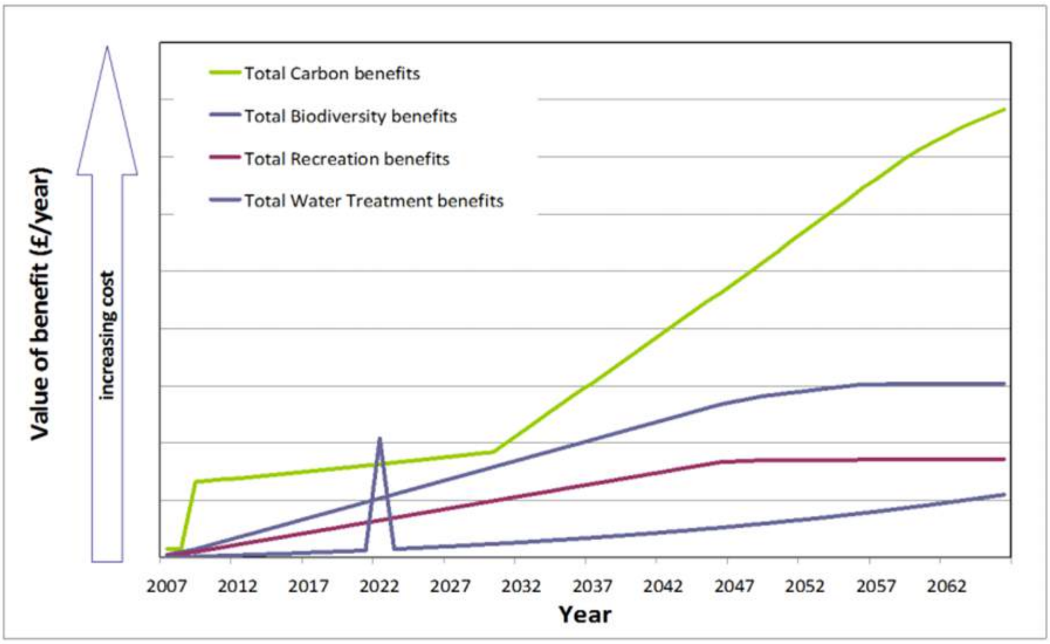


Figure 2-11 Value of anticipated co-benefits arising from SCaMP

2.1.147 Similarly, Yorkshire Water and Natural England assessed how land management scenarios in the Keighley and Watersheddles catchment in the South Pennines would affect the provision of a range of benefits. The study found that restoring and re-wetting upland blanket bog over a relatively small area (around 3,000ha) could deliver estimated net benefits of £6.3m over a 25 year period. Overall, the benefits from increased carbon sequestration, improvements in wildlife and reductions in water treatment costs exceeded restoration costs by a ratio of 3:1. The study also looked at the costs and benefits of further deterioration in the quality and extent of the blanket bog. Under this scenario, the analysis found that the benefits of preventing further declines exceeded costs by over 5:1. Moreover, the study found that the ratio of the present value of the improvements across the three services (carbon sequestration, biodiversity and water quality) was suggesting that carbon sequestration values could potentially be worth more than double the value of water quality improvements, and double the value of biodiversity.

2.1.148 Drawing on this and other evidence, the Third State of Natural Capital Report (Natural Capital Committee 2014) found a strong economic case to support investments in activities aimed at rewetting certain areas of peatland which would lead to significant benefits beyond carbon storage including improved water quality and habitat for wildlife. By way of example, the authors estimated that improvements on around 140,000 ha of upland peatland could deliver benefits (in net present value terms) of approximately £560m over 40 years. However, this estimate only took into account the avoided loss of carbon and is therefore likely to significantly underestimate the total value of benefits which would include wildlife and water treatment cost savings. However, as the report noted, these additional benefits vary significantly across catchments and cannot be reliably valued without more detailed spatial analysis.

2.1.149 Continued monitoring of the effects of upland land management activities and of the costs and benefits involved provide useful information to United Utilities (and other water companies who are undertaking similar programmes) and their regulators when faced with decisions about how and where to invest in order to enhance the reliability and quality of water supplies without unduly raising the costs to their customers. Similarly, this information will also be useful to land managers and those responsible for supporting catchment management and countryside stewardship initiatives.

Specific pollutants, priority substances, and 'other' chemical pollutants

2.1.150 The UK's monitoring programme for pollutants is split into two main target groups: 1). Those pollutants described in Annex VIII of the WFD, known as 'Specific Pollutants', and 2). Those pollutants listed in Annex of the WFD, known as 'Priority Substances'. The former contribute towards the assessment of ecological status under the WFD, while the latter contribute towards the assessment of chemical status. The levels of these pollutants must be below the relevant Environmental Quality Standards (EQS), as per the EU Directive on EQS (EU 2008). The 'Priority substances' (n=33) are standard across the EU, while the constituents of (and EQS for) the 'Specific Pollutants' list are suggested and reviewed by UKTAG, and represent those pollutants of national or local concern. The last review of Specific Pollutants was completed in 2013 (UKTAG 2013b).

Description of the problem

2.1.151 **Most WFD failures for these pollutants occur in urban or ex-mining contexts, with multiple, diverse origins.** These range from point source domestic and industrial, to diffuse pollution from agriculture. Given that many of the chemicals defined as 'Specific pollutants' and 'Priority substances' have numerous uses, the potential contexts in which the combined group of chemicals are applied are too numerous to describe exhaustively here, but include most primary (e.g. arable farming) and secondary (e.g. pharmaceutical) industries, and domestic uses. Most of the primary industry sources are diffuse, while secondary industry and domestic tend to be point sources. CSOs and stormwater runoff in towns and cities, leaching from landfill sites (where there is no impermeable base layer), and leaching from ex-mining sites can also result in point source releases of these pollutants. Drainage of heavy metals from mine sites can also reduce dissolved oxygen levels, as when they precipitate, they remove oxygen from the water.

2.1.152 By definition, the chemicals defined under the WFD are monitored and legislated for because they cause harm to the water environment, with direct impacts mainly due to their toxicity. Here again, the number and variety of chemicals involved preclude a full description of their impacts, but these impacts particularly manifest where other exacerbating factors occur, such as: bioaccumulation, which raises the concentration of pollutants in the biota to more harmful or lethal levels (where the chemical involved is hard to metabolise); low pH, which increases the levels of soluble toxic metals (such as aluminium); or low flows, which increases their concentration in the water body. Now that the constituents and standards for both groups of pollutant have been set, monitoring under the new regime will facilitate an improved understanding of the contexts in which safe levels are most often exceeded.

Trends

Present day trend: ─ (medium confidence).

Future trend: ─ (low confidence).

2.1.153 **Specific pollutants are responsible for 9.8% of the failures listed in the England and Wales data.** WFD Sampling for chemical pollutants does not occur in every water body of the UK, and monitoring is prioritised for areas in which concern is greater (e.g. urban areas). The failures under specific pollutants mostly arise from levels of the heavy metals zinc and copper, which account for 4.3% and 3.9% of the total failures in England and Wales (Table 2-10). The water environment in Wales is less affected by pesticide use; the 'Reasons for not achieving good' dataset contains a solitary failure attributed to Cypermethrin (although see Environment Agency 2010). The remaining Specific Pollutant failures in Wales were associated with Copper (n=24), Iron (n=4) and Zinc (n=76). Ammonia is listed separately to other specific pollutants, and accounts for 6.2% of total failures.

2.1.154 The SEPA data cite specific pollutants as responsible for 1.8% of the failures in Scotland, although the pollutant involved in these cases is not identified.

Table 2-10 The species of specific pollutant behind WFD failures in England and Wales

Species	Count
2,4-Dichlorophenol	3
2,4-Dichlorophenoxyacetic acid	2
Copper	245
Cyanide	1
Cypermethrin	28
Diazinon	10
Iron	47
Mecoprop	1
Permethrin	9
Trichloroethylene	2
Zinc	273

2.1.155 **Priority substances are responsible for 3.0% of WFD failures in England and Wales, and 0.5% of failures in Scotland.** In both datasets the chemical involved is not identified. The constituency of the 'Priority substances' group has recently changed, and as a result, for a number of this revised group (e.g. fluoranthene, mercury), compliance will not be assessed until Cycle 2 of the WFD (2016-2021, Defra 2014a). Early indications are that overall compliance will fall following the assimilation of these data into the monitoring database.

2.1.156 FERA conduct an annual survey of pesticide use which reports on nationwide trends, which has revealed that the total weight of pesticide (all types) applied in the UK is declining, but the total area over which it is applied is increasing (Figure 2-12). It is difficult to get an idea of the potential impact of these changes on the water environment as one species of chemical may be more potent or detrimental than another; more persistent; more susceptible to leaching; or more harmful if applied at higher concentrations. The use of two or more pesticides may also result in synergistic impacts that outweigh their estimated impacts when used in isolation. It is probably the case that pesticides have become more potent since 1990. Clearly any future changes to the trends in all pesticide use will reflect changes in farming and future regulation (see also 'Adaptation' section below).

2.1.157 The EU imposed a ban on three types (clothianidin, imidacloprid and thiametoxam) of neonicotinoid insecticides, due to their impact on wild bees (Rundlöf et al. 2015) and other non-target invertebrates (see Pisa et al. 2015 for a review). This ban was implemented in 2013, and will be reviewed at the end of 2015. A recent emergency derogation of this ban by the UK Government will see the time-limited (120 days maximum) use of clothianidin and thiamethoxam in the areas of oil seed rape cultivation that have been worst affected by pests (likely to be the East of England). This use will be restricted to 5% of the cropped area.

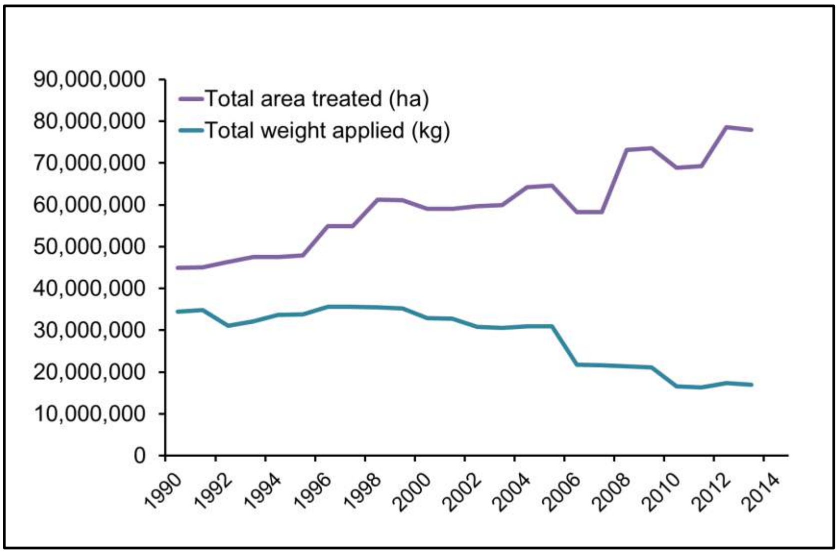


Figure 2-12 Trends in pesticide use compiled by FERA (2014)

Impacts of climate change

2.1.158 The impact of climate change on the risk from these pollutants will be dependent on changes in precipitation at particular times of year, although rising temperatures could mitigate the risk by increasing degradation rates (Table 2-11).

2.1.159 As chemical pollutants are a particular problem in urban areas, an increase in storminess would result in more CSO discharges and thus the transport of more pollutants into the water environment (particularly zinc, Seidl et al. 1998). This change would also mobilise such pollutants that may otherwise have remained inert in sediment beds, i.e. those that are not presently considered bioavailable. For example, a study of a CSO in Paris reported the presence of 14 Priority Substances in water downstream of a CSO discharge; the concentrations of 8 of these pollutants were above WFD maximum levels (Gasperi et al. 2009). These contaminants can lead to acute and chronic toxic impacts on aquatic life, while it also acts as a continuing source of persistent bioaccumulative toxic chemicals (US EPA 2011).

Table 2-11 Overview of climate change impacts on specific pollutants, priority substances and other chemicals

Direction of impact	Climate variable		Mechanism(s)
	Present trend (confidence)	Future trend (confidence)	
Exacerbating	Summer precipitation		Concentration.
	↓ (Low)	↓ (Medium)	
	Storminess		Direct transport; Indirect via CSOs.
	- (High)	↑ (Medium)	
Mitigating	Winter precipitation		Increased dilution.
	↑ (Low)	↑ (Low)	
	Temperature		Volatilisation; Degradation.
	↑ (High)	↑ (High)	

2.1.160 Unlike the impact of projected changes to precipitation on nutrient enrichment (Table 2-4), contrasting trends in winter and summer precipitation are likely to have contrasting impacts on pollution via these chemical groups. More precipitation at any time of year will lead to reduced concentrations of these pollutants, whereas less precipitation will result in increased concentrations of these pollutants. The net impact of either direction of change will depend on how these changes affect the hydrological regime and flows (both surface and groundwater) proximate to, and upstream of, pollution sources.

2.1.161 Expected increases in temperature will reduce the persistence of pollutants in the environment, via degradation, and thus reduce residence times. Higher temperatures will also make volatilisation more likely in those chemicals prone to it, such as the herbicide 2,4-Dichlorophenoxyacetic acid (commonly referred to as 2,4-D, and a 'Specific pollutant'). The net outcome of the volatilisation impact may not necessarily be positive, as the newly gaseous chemicals can damage to non-target wildlife or crops downwind.

Adaptation

2.1.162 Some of the measures to reduce the impact of specific pollutants, priority substances and pesticides are shared with those targeting nutrient pollution, such as SUDS and diffuse pollution control. However, vegetation buffers, biobeds and biofilters can also be employed, while the wider rollout of UV treatment would reduce the need for chlorine at wastewater treatment works.

2.1.163 Measures to reduce diffuse pollution from agriculture, although aimed at reducing fertiliser runoff, will also reduce the runoff of pesticides. This will improve their residence time in the intended application area, and thus has the potential to save the farmer money by increasing the intervals between re-applications. Increased deployment of more sophisticated alternatives to prophylactic (preventative) application of pesticides in agriculture, such as Integrated Pest Management (IPM) strategies, would see usage fall as pesticide would only be applied if, where and when it is needed (Furlan and Kreutzweiser 2015). This approach would also lower the likelihood of the target pests developing resistance to the pesticide being applied, improving food security (Szendrei et al. 2012). More successes in the application of biocontrol techniques in Europe could also see their wider application in agriculture to target problem pest species.

2.1.164 Pesticide release into the environment via point sources (e.g. in pesticide handling or washdown areas) can also be attenuated using biobeds or biofilters (Fogg et al. 2003). These technologies target potential point sources of pesticide release into the environment by providing a barrier between the source and the soil, trapping the pesticide in a 'biomix' of soil, compost and straw. This biomix contains the bacteria required to break the pesticide down (Henriksen et al. 2003). The resulting leachate (from the bed or filter to the soil) contains very low concentrations of pesticide. The Environment Agency provides a waste exemption for 15,000 Litres of non-hazardous pesticide washings to be treated in this manner, per year (Environment Agency 2014).

2.1.165 Vegetation buffer strips can also be applied to reduce pesticide loss from agricultural land. Nisbet et al. (2011) suggest three potential uses, namely: 1) Planting or preservation of windbreaks of more than 1m in height around fields in which pesticide is being sprayed, which has been shown to reduce the airborne 'drift' of pesticide to other fields by 25-90%, depending on the amount of foliage in the buffer (Brown et al. 2007); 2) Riparian buffer strips to intercept pesticide entry into watercourses, although the evidence for their efficacy is equivocal (Reichenberger et al. 2007); 3) Constructing wooded wetlands to treat contaminated areas, and here findings from the EU ArtWET project suggest that 60-90% of the residual pesticide can be removed (depending on the type of pesticide and its method of delivery, Nisbet et al. 2011). Due to differences in the experimental design and assumptions behind studies investigating the efficacy of these buffers, there remains uncertainty over the degree to which these techniques are applicable in a general sense, and further research is thus required.

2.1.166 Porous pavements, swales, marshland, and other features of SUDS increase the residence time of the pollutants in the soil, thereby exposing them to a higher level of degradation before they reach watercourses. The beneficial impact of SUDS can also be enhanced by the promotion of domestic and industrial practices that reduce pollutant runoff, such as sweeping hard surfaces where the chemical is being used, or covering them with a canopy or roof. The appropriate disposal of polluting chemicals (e.g. detergents or antifreeze) can also be encouraged.

2.1.167 Ultraviolet (UV) disinfection is deployed at sewage treatment works as a means of complying with the Shellfish and Bathing Water Directives (EU 2006a,b). UV targets microorganisms (such as Cryptosporidium) by inactivating them, rendering them unable to reproduce or infect. Use of UV in treatment processes has been increasing since an amendment to the Water Supply (Water Quality) Regulations for England and Wales; specifically, a regulated organism or substance can now be rendered 'harmless', as opposed to be completely removed, during the treatment process (CIWEM 2010). Because the water authorities offer flexibility over how compliance with the EU Directives is achieved (e.g. Environment Agency 2012), the use of UV has the benefit of reducing the outflow of chemicals such as Chlorine (a 'Specific pollutant') that would otherwise be used in alternative, secondary processes. UV treatment is also increasingly used as an alternative to additional storm storage tank capacity, which would otherwise be required for compliance with these Directives (e.g. Cog Moors, South Wales, Barcock and Scanell 2010). Thus UV also mitigates the health risks associated with CSOs. At the time of writing, statistics describing the use of UV across the UK were not available.

2.1.168 In a similar manner to actions targeting nutrient pollution, the use of suction dredging could be considered for reservoirs of these chemicals in sediment beds for problem areas, especially where more intense rainfall under climate change results in their mobilisation. The WFD standard for some specific pollutants has been recently amended to specify that they must be 'bioavailable' (Defra 2014a), and thus such long-term reservoirs may not result in WFD failures in the future. Hence, although the persistent properties of many such pollutants will mean that they remain a risk for a number of years after their deposition, measures geared towards WFD objectives will likely be focussed on more cost-effective approaches.

Over-abstraction and saline intrusion

Description of the problem

2.1.169 Abstraction refers to the taking of water from any source, although in practice these sources are mostly groundwater stores and river channels (canals, reservoirs and lakes are also used). Here, we take 'over-abstraction' to be abstraction that results in harm to the water environment, rather than that which is considered to be over-licenced under Defra's licensing regime (Defra 2013a).

2.1.170 Saline intrusion is the movement of seawater into freshwater aquifers (groundwater). Although some intrusions are natural, here we are broadly referring to intrusions where abstraction is the cause.

2.1.171 Negative impacts of abstraction can include all the negative impacts associated with low flows, such as high river temperatures, fish stress and mortality, reductions in invertebrate density, and increased concentrations of other pollutants. Thus abstraction may also have indirect impacts on other sources of pressure which may be recorded as the primary driver of harm. The direct impact on flows may be captured by WFD measurements of the hydrological regime itself.

2.1.172 Abstraction is licensed (and *de facto* managed) by the Environment Agency in England, NRW in Wales, SEPA in Scotland, and DOENI in Northern Ireland. The existing licensing system has been largely unaltered since the mid-1960s, although the introduction of Catchment Abstraction Management Strategies (CAMS, Environment Agency 2002) brought a standardised approach to determining water availability, upon which decisions on new licences are made. This was the first such approach in Europe at the time. The present regime captures abstractions of 20m^3 per day or more, although changes to the framework for licensing were recently put out to consultation, and an improved link between water availability and abstraction was proposed, particularly at times of low flow (Defra 2013b, see also 'Adaptation' below).

2.1.173 Saline intrusion can result from over-abstractions from groundwater supply (via pumps, boreholes or wells), wherein the hydraulic gradient from the land to the sea is weakened, and sometimes reversed, by the removal of freshwater. This removal can also be on a more permanent basis, for land drainage (e.g. Norfolk Broads, Holman and Hiscock 1998). Because sea water is more dense, the intrusion will (at first) occur in the lower parts of the aquifer, with the freshwater-seawater boundary (or interface) moving landwards. Eventually the abstractor will experience an increase in the chlorine content of the retrieved water supply, which reduces the motivation to extract the water and can act as a self-regulating feedback.

Trends (abstraction)

Present day trend: ─ (medium confidence).

Future trend (demand only): ↑ (low confidence).

2.1.174 The above trends refer only to rates of abstraction, and not over-abstraction as we define it (see above). Trends in the latter (both present, and future projected) would be difficult to establish as the duration and volumes of abstraction vary over far shorter timescales than current monitoring regimes for the water environment can capture. Note that our future trend estimate represents anticipated future demand, and does not take into account any future measures to manage demand or improve water efficiency.

2.1.175 The treatment of abstraction within the above WFD water quality datasets differs by country, and it is difficult to establish a national trend from these data. Abstraction is cited as a source of 3% of the failures in the 'Reasons for Not Achieving Good' dataset for England and Wales, but of 21.5% of failures in the Scottish dataset. These variant estimates possibly arise from differences in the assessment criteria, with Scotland for example including abstraction for renewable electricity production in its total for abstraction failures. The discrepancy could also be due to differences in previous approaches to regulation.

2.1.176 The WFD 'Reasons for Not Achieving Good' dataset for England and Wales mainly captures abstractions by the water industry and agriculture. Other estimates of the impact of abstraction on WFD performance in England and Wales have also been compiled. In an analysis of present day availability of water, the Environment Agency and Ofwat (2011) estimated that rates of abstraction are putting good ecological status at risk in over 1,000 water bodies in England and Wales (11% of the total, Environment Agency and Ofwat 2011). A more recent estimate (Environment Agency 2013a) suggested that 13% of river water bodies in England and 4% in Wales are failing to support good ecological status due to abstraction. The same document suggested that 42% of groundwater water bodies in England and 6% in Wales are also failing to support good ecological status for this reason.

2.1.177 The total amount of water abstracted each year is estimated to be 14.1 billion m^3 in England and Wales, and has remained within the range 12 -15 billion m^3 since the year 2000 (Defra 2015b). Recent overall demand for water has also shown no trend (Watts et al. 2013), although there has been a 2% per annum increase in demand for irrigation water (Weatherhead and Howden 2009). The major areas of arable crop growth requiring irrigation tend to be the areas more likely to be supplied by groundwater aquifers, and this irrigation also tends to be required at times of year (mid to late summer) where supplies are at their lowest. Demand for irrigation is also likely to increase under climate change (see 'Impacts of climate change' below).

2.1.178 Using the Water Exploitation Index (WEI+), the Environment Agency and Natural Resources Wales (2013) assessed all catchments in England and Wales to determine the level of stress placed on the water environment from the use of water through abstraction, discharge and management of storage (Figure 2-13). Many of the areas under 'serious' stress were in the South and East of England. In the same analysis, the Environment Agency and Natural Resources Wales (2013) found that of the 24 water company areas across England and Wales, nine had water bodies at high risk of environmental impacts as a result of overexploitation from abstraction.

2.1.179 It is unclear how projected increases in population will affect abstraction rates. The population of the UK was estimated to be 64.1 million in mid-2014, and the ONS (2015) project this to increase to 73.3 million by mid-2027. This growth is expected to occur across all countries of the UK, but particularly so in England (0.6% growth rate), as opposed to Northern Ireland (0.4%), Wales (0.3%) and Scotland (0.3%). The Environment Agency and Ofwat (2011) estimated future demand for water in 2050 to be between 15% less and 30% more than it is today. However, the outcomes of the 2014 price review (PR14) show that only 2 out of 18 water companies will be increasing the amount of water they abstract for public supply over the price control period (2015-2020), despite increases in population.

2.1.180 A total of 13 failures were attributed to saline intrusion in the England and Wales WFD data, and 12 failures in Scotland, with negative impacts on invertebrates and polybenthos described. The frequency of saline intrusions is likely to increase given sea level rise (present, and future projected), and continued vertical land movement, which will continue to 'lower' the relative elevations of those regions containing coastal freshwater aquifers (and thus more prone to saline intrusion), i.e. South-East England.

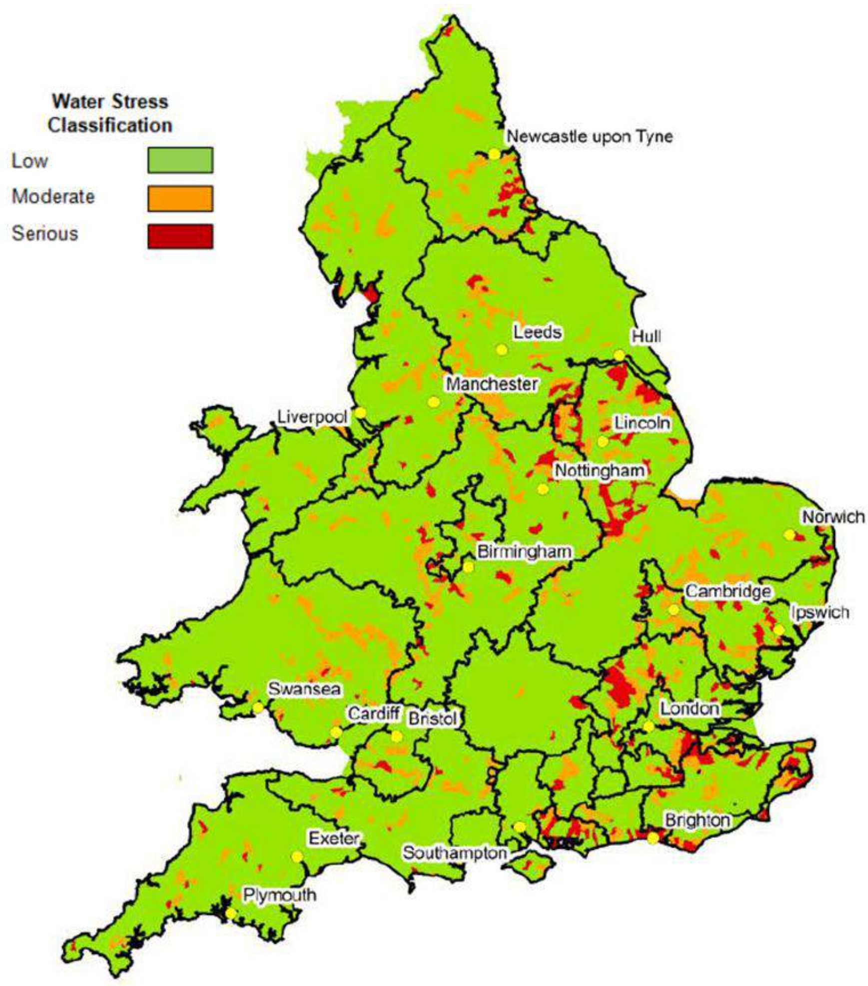


Figure 2-13 Water stress in England and Wales

(Source: Environment Agency and Natural Resources Wales 2013)

Impacts of climate change

2.1.181 Similar to nutrient enrichment, direct human impacts (in terms of usage) will be the dominant control on the risk of over-abstraction, although many of the present and future projected trends in climate change exacerbate the problem (Table 2-12). Although the direction of travel with respect to mean temperature and extreme hot conditions is obvious, much will also depend on future precipitation levels under warming, which are uncertain (Murphy et al. 2009). Summertime is likely to see the culmination of adverse impacts. Hot summers are characteristically low in rainfall, which reduces groundwater recharge and the availability of surface water, while resulting in lower flows. A higher frequency of either low flows or drought conditions would reduce dissolved oxygen concentrations, and thus could have adverse impacts on wildlife (Cox and Whitehead 2009).

Table 2-12 Overview of climate change impacts on over-abstraction and saline intrusion

Direction of impact	Climate variable		Mechanism(s)
	Present trend (confidence)	Future trend (confidence)	
Exacerbating	Sea level rise		Thermal expansion of the oceans will continue to drive sea level rise and increase salt load.
	↑ (High)	↑ (High)	
	Summer precipitation		Decreases in summer precipitation will reduce groundwater recharge and increase rates of abstraction
	↓ (Low)	↓ (Medium)	
	Temperature		Higher temperatures will increase demand for water, particularly during the summer months.
	↑ (High)	↑ (High)	
	Drought		More droughts would increase abstraction as other (surface) sources of water dry up.
	↑ (Low)	↑ (Low)	
Mitigating	Winter precipitation		More winter precipitation (if not intense) would increase groundwater recharge and decrease the risk of over-abstraction.
	↑ (Low)	↑ (Low)	

2.1.182 Sea level rise will result in increases in salt load as the freshwater-saltwater interface is pushed inland (Table 2-12). Despite the widespread focus on adapting to coastal inundations driven by sea-level rise, a recent global meta-analysis concluded that many coastal aquifers (including those of the UK) are far more vulnerable to this impact (Ferguson and Gleeson 2012), emphasising the need for more policy attention dedicated to water management. Agricultural users may thus be adversely affected by rises in the salt content of groundwater boreholes they use for water supply.

2.1.183 The impact of future climate change on groundwater supplies and recharge is uncertain. The shift in rainfall amounts, broadly from summer to winter, could in theory benefit some groundwater supplies currently prone to drought, via increased recharge (Marsh et al. 2013). However, few studies have tested this idea. A study of the chalk aquifer of the Marlborough and Berkshire Downs and South-west Chilterns (MABSWE) showed that the projected increases in winter rainfall (described above) could bring about an increase in spring baseflow (medium-high emissions scenario, A2, 2080s) which would increase the risk of groundwater flooding events at t++hat time of year (Jackson et al. 2011). But the same study also concluded (more strongly) that autumn baseflow could be reduced, and thus, net recharge will be largely unaffected. There is further uncertainty over the extent to which 'fast' recharge events (via macropores and fissures) would increase recharge if the intensity of rainfall were to increase (Arnell et al. 2015).

2.1.184 Drought (meteorological and agricultural) risk will increase in summertime periods, posing risks to the agricultural sector and the public water supply. Low flows (Q95) are projected to decrease by 20–40% nationwide by the 2050s (A1B or medium emissions scenario, FFGWL project, Prudhomme et al. 2012). Using an example from the South of England, Prudhomme et al. (2012) showed how the catchment level changes in FFGWL can be used to calculate how future levels of abstraction may have to be modified to meet WFD requirements- with a ratio of Q95 to abstraction commonly used to assess the level of risk. Given that many of the projected decreases in low flow volume for the 2050s exceed the threshold that the authors suggest is 'dangerous' (15%), abstraction in groundwater-fed catchments will likely have to be reduced in the summer.

2.1.185 A study by Environment Agency and Ofwat (2011) showed that a 15% reduction in flows would mean that large areas of the South-west, Western Wales and the North of England would change from having water available (for new licences) 95% of the time to having water available less than 70% of the time (Medium emissions scenario, 2050s). This modelling also suggested that some catchments could have unmet demand in the future; more often than not these are also the areas facing the largest reductions in flow as a result of climate change (Environment Agency and Ofwat 2011).

2.1.186 The difficulty that climate models have in reproducing multi-annual rainfall deficits could mean that these more persistent types of risk are underestimated in assessments of climate change impacts (Prudhomme et al. 2012). To assess the worst case scenario, the CCRA2 Evidence Project D derived a 'High++' scenario for low rainfall which would see summer rainfall deficits running at 60%, or 20% for longer durations (multi-annual). The impact of these events would be comparable to the worst drought events on record (1975-76 for short duration, 2010-12 for multi-annual, Wade et al. 2015).

2.1.187 **Demand from irrigation has been projected to increase by a global average of 1–3% by the 2020s and 2–7% by the 2070s** (range stated across IPCC SRES scenarios, IPCC 2008). Projected future changes to the variability of precipitation can also affect agricultural yields, and the negative impact of this variability on yields can be on a par with the impact of projected reductions in the total amount of precipitation (Eheart and Tornil 1999). This research highlights the importance of water storage adaptations that will 'dampen' this variability, and provide more water in reserve should it be required (See below).

2.1.188 Another CCRA2 Evidence Project (Project B) built on the data and models behind the FFGWL project to generate projections of water availability for the UK. Under high levels of climate and population change, the projections revealed that most of the UK (particularly South-East England) would be in supply-demand deficit by the 2050s. The project also found that, under low levels of climate and population change, significant deficits would still be evident.

Adaptation

2.1.189 The principal means of limiting the impact of over-abstraction is via abstraction licensing, which is currently undergoing reform. Other gains are possible by improving water storage, soil management and water efficiency.

2.1.190 A commitment to abstraction reform in England and Wales was set out in the Water White Paper of 2011, and a public consultation was launched in 2013 (Defra 2013a). A summary of responses has been published (Defra 2014b), and it will be for Government to draft and pass legislation to put any changes into effect. Environmental groups have concerns that old (inappropriately distributed) licences will be grandfathered to historic users by the new regimes, while farming groups are concerned that they will lose (or be outbid for) their access to water. Improving the tradability of abstraction rights should ensure that the system responds to need, but better monitoring data will be required to ensure they are also matched to availability.

2.1.191 The Environment Agency and Natural Resource Wales operate the Restoring Sustainable Abstraction programme, targeting areas where performance under the WFD may be at risk from over-abstraction. Schemes involving 600 licences are currently in operation (Environment Agency 2013a), out of a total of approximately 25,000 licences for abstraction (and impoundment) that are 'live' across England and Wales (Environment Agency 2009). Remedial actions under these schemes can involve changes to the location, volume or timing of abstraction, the restoration of physical processes (e.g. by improvements to river channels), or the identification of alternatives to the abstraction. Licence changes may be voluntary or compulsory.

2.1.192 The ability of water authorities to predict which summers will present the most problems regarding abstraction is limited by the fact that low flow levels predominantly respond to the precipitation amount in that summer (i.e. there is little or no lead time, Wilby and Harris, 2006). Thus adaptations for water management focus on increasing the capacity of public and private water supplies to respond to all plausible scenarios for rainfall, both under present day conditions and with climate change.

2.1.193 In this regard, a recent NFU survey found that 41% of abstraction licensees in East Anglia are planning to invest in reservoir capacity or have already done so (NFU 2015). The expense of constructing reservoirs can sometimes be prohibitive (Global Food Security 2015), but proposals that would allow users to top up reservoirs during high flows in the summer could make the decision to construct them more attractive (Defra 2013a). An increased level of water security for farmers would also permit them to plan cropping more effectively, and embark on more profitable sectors such as salad crops or vegetables (that would otherwise be too risky to take on). This can also be achieved by measures focussed on the demand side, such as irrigation scheduling (e.g. at night), water recycling, and water sharing (voluntary or otherwise).

2.1.194 In Wales there is an east-west divide in the commitment of water resources, such that regions in the east of the country are often heavily committed (water available for new abstractions less than 30% of the time), while regions in the west tend to suffer less pressure (water available for new abstractions over 90% of the time, NRW 2014). Thus although nationally the existing provision of water storage may seem sufficient, the east of Wales is likely to require more reservoirs given projected climate change.

2.1.195 Improved soil management could also have the potential to reduce surface water runoff in winter, and thus increase winter recharge to groundwater stores (Palmer and Smith 2013). Hess et al. (2010) found substantial potential across England and Wales for improvements in soil condition to reduce surface water runoff during intense rainfall events, particularly in drier regions (which will be more reliant on abstraction). In a separate study of soil degradation, 38% of sites surveyed in the South-west had 'high' or 'severe' levels of soil structural degradation (Palmer and Smith 2013). This degradation was most evident where late-harvested crops such as maize and potatoes were grown, or winter cereals planted in late autumn. There may also therefore be potential to reduce surface water runoff by changes in the types of crop grown in sensitive areas (e.g. steep slopes, or inappropriate soil types), or changes to the timing of cropping. If implemented, these improvements to soil management could also bring flood alleviation benefits, reduced nutrient transport to water bodies and improved performance under the Water Framework Directive. Potential changes to the national approach to soil management will be outlined in the new soil strategy for England. The evidence gathering phase for this strategy will end in 2016, after which a revised strategy will be produced (CCC 2015).

2.1.196 Climate change will alter patterns of consumer demand for drinking water, and is likely to increase demand overall (CCC ASC 2010). Measures to address this include demand management, variable consumer tariffs, the construction of facilities for water re-use, and improving the connectedness of regional water networks. The Ofwat water efficiency targets for the water companies are to be reviewed this year (2015). Wider roll-out of water meters in households (almost half of households, CCC 2015) will result in lower water usage, as awareness improves.

Valuation case study #4: Environmental flows

2.1.197 The IUCN defines environmental flows as the "water regime provided within a river, wetland or coastal zone to maintain ecosystems and their benefits where there are competing water uses and where flows are regulated" (Dyson et al. 2003). In other words, these flows refer to the volume of water that is not used by humans for consumption or production), but supports the ecological functioning of aquatic ecosystems, which, in turn, support other ecosystem services such as recreation and habitat for wildlife.

2.1.198 This definition implies that, to be beneficial, environmental flows must comprise the right volume and quality of water, available at the right time to maintain ecosystems. The combination of these elements (quantity, quality and time) means that it is difficult to quantify the impacts climate change may have on environmental flows. A simplified impact pathway is set out in Figure 2-14.

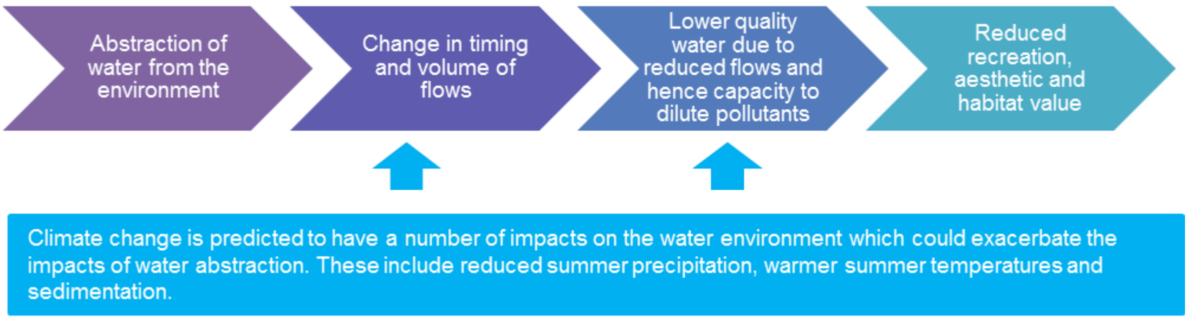


Figure 2-14 Impact pathway for the impact of water abstractions on environmental flows

2.1.199 Given the complexity of the relationship between the quantity, quality, and timing of environmental flows and the impact on the beneficiaries of the ecosystem services provided, it is difficult to develop a robust impact pathway that could be used to quantify and value the impact of climate change on environmental flows. Furthermore, we were unable to identify any studies that looked specifically at the relationship between changes in water quality linked to environmental flows and associated benefits to aquatic ecology and other instream beneficiaries. The examples provided below therefore document approaches to valuing changes (improvements) in water quality regardless of the driver of change and are also simplifications of the interplay between the elements that contribute to environmental flows (quantity, quality and timing). Nevertheless, they provide an indication of the likely order of magnitude of the costs and benefits of the changes in water quality that may be experienced as a result of changes in the timing and volume of flows and the capacity of these flows to dilute concentrations of pollutants.

The Environment Agency Approach

2.1.200 The Environment Agency's Water Appraisal Guidance (WAG) describes an approach to identifying and valuing the significance of impacts on ecosystem services that is consistent with the Treasury Green Book. As part of this approach, the Environment Agency has compiled estimates of the benefits of improvements in water quality per kilometre for the main river basins in England and Wales, accounting for population densities and the availability of alternative sites for recreation (Environment Agency 2013b).

2.1.201 The approach focuses on estimating the value of three freshwater ecosystem services: biodiversity (in terms of fish and other aquatic life); aesthetic quality (visual, clarity, smell, insects); and recreation (suitability for relaxing, in stream and near stream activities). Average benefits were found to be £15.6/km, £18.6/km and £34.2/km for improvements that lift water quality from low to medium, from medium to high, and from low to high respectively. While these estimates do not extend to river basins in Scotland and Northern Ireland, the underlying methodology could be adapted to derive estimates of changes in water quality in these countries.

2.1.202 The Environment Agency approach draws on earlier work by Metcalfe *et al.* (2012) which was undertaken on behalf of Defra. A key feature of the model developed by Metcalfe et al. is that it can be used to value individual localised improvements as a function of the size of the area improved, the qualitative scope of improvement (low to medium, medium to high or low to high), and the population density of the area surrounding the water body. The waterbody valuation function developed by Metcalfe et al., is given by:

$$-(ps_{LowLO}^{CV} + qs_{LowNO}^{CV})$$

2.1.203 Where the parameter *p* is a local scalar equal to the population living within 30 miles of the water body in question divided by 1% of the area of a 30-mile radius circle, and *q* is a national scalar equal to the national population, divided by 1% of the area of the country, including coastal areas. s^{CV}_{LowLO} and s^{CV}_{LowNO} represent scaled contingent valuation estimates (at a 95% confidence interval) of WTP for an improvement in water quality from low (bad WFD status) to medium (moderate or poor WFD status) at the local and national level respectively. The function thus effectively calculates the aggregate value of an improvement in water quality (from bad to moderate status) taking into account the preferences of both local and national populations who hold value for the waterbody in question.

2.1.204 The estimates (from Metcalfe et al., 2012) presented in Table 2-13 show the low-scaled and high-scaled WTP estimates (in terms of £ per household per year and at a 3% discount rate) for percentage improvements in local high and low quality, and national high and low quality over an 8-year improvement programme. The low and high-scaled estimates were derived using two different survey elicitation methods: the low scaled estimates were derived using a payment card contingent valuation approach while the high-scaled estimates were derived using a dichotomous choice contingent valuation approach. The differences in these approaches is described in detail in Carson and Groves (2007, 2011) and are not repeated here.

Table 2-13 Scaled WTP estimates from marginal changes in current status

	Low-Scaled WTP ($£hh^{-1}yr^{-1}$)	High-Scaled WTP ($£hh^{-1}yr^{-1}$)
	Discount rate (3%)	Discount rate (3%)
S^{CV}_{HighLO}	0.16	0.41
S^{CV}_{LowLO}	-0.11	-0.28
S^{CV}_{HighNO}	0.20	0.51
S^{CV}_{LowNO}	-0.16	-0.41

(Source: Metcalfe et al. 2012).

2.1.205 The methodology set out by Metcalfe et al. results in an estimate of WTP per household per year for an improvement in water quality from a number of assumed starting points (i.e. low and medium water quality). As such, the resulting estimates can be considered to reflect the value that households place on achieving a given level of water quality.

2.1.206 However, an important limitation of this approach is that it examines only WTP for improvements in water quality. It does not tell us about WTP to avoid the loss of nonmarket benefits if there were considerably lower standards of water quality in UK freshwaters, other than suggesting that these are likely to be very significant.

2.1.207 It is incorrect to assume that WTP for an improvement in water quality is equivalent to WTP to avoid deterioration in water quality or, conversely, WTA compensation for a deterioration of water quality. Research into the difference between WTP and WTA estimates (Horowitz and McConnell 2002) suggests that WTP estimates are typically less than WTA estimates. The Horowitz and McConnell meta-study found that the mean ratio of WTA to WTP across a sample of studies was 7.17, while the median was 2.60. In the absence of estimates for WTP to avoid a deterioration in water quality, the estimates derived from the Environment Agency approach could therefore be multiplied by a factor of 2.6 (and 7.17 for the purposes of sensitivity testing).

2.1.208 Thus, with some caveats, the approach provided by the Environment Agency could be used to quantify the value of changes in environmental flows in terms of their ability to provide non-consumptive ecosystem services. However, further work would need to be undertaken to disentangle the complex range of pathways through which climate change can impact on the volume, quality, and timing of environmental flows.

The REFRESH project

2.1.209 The REFRESH Project on Adaptive strategies to Mitigate the Impacts of Climate Change on European Freshwater Ecosystems is an EU funded project under the Seventh Framework Program[3]. The key objective of REFRESH is to develop a framework to enable water managers to design cost-effective restoration programmes for freshwater ecosystems that take account of the expected future impacts of climate change and land-use. One of the key objectives of the project is to develop and demonstrate effective methodologies to assess the cost-effectiveness of alternative adaptation/mitigation strategies in freshwaters.

2.1.210 Within the REFRESH project, Martin-Ortega et al. (2014) investigated the cost-effectiveness of remediating strategies within the Thame sub-catchment of the Thames Basin to achieve compliance under the WFD and Habitats Directive. In particular, the research assessed the (dis)proportionality principle[4]. Table 2-14 shows the initial investment and duration of ongoing annual costs of measures to achieve GES in the Thame sub-catchment. The initial investment costs represent the capital costs, where initial costs are the up-front costs required to achieve GES and annual costs refer to operational and maintenance costs.

Table 2-14 Costs and lifetime of measures that achieve GES in Thame sub-catchment

Measure	Initial investment costs (GBP in year 0)	Annual costs (GBP per year)	Lifetime (years)
Establish riparian buffer strips (10 meters width)	-	54,000	Continuous
Reduce 20% P fertilizer reduction applied across all crop land	-	561,600	Continuous
Adopt minimum tillage systems (50% combinable crops)	-	1,404,000	Continuous
Establish and maintain constructed wetlands (10% of all arable and grassland)	992,200	313,365	10
Establish winter cover crops	-	473,850	Continuous

(Source: Martin-Ortega et al. 2014)

[3] See http://www.refresh.ucl.ac.uk/about/background

[4] The Water Framework Directive (WFD) aims to deliver good ecological status (GES) for Europe's waters. It prescribes the use of economic principles, such as derogation from GES on grounds of disproportionate costs of mitigation.

2.1.211 The non-market benefits of improving water quality in the Thame sub-catchment were estimated based on the findings contained within a report by NERA and Accent (2007) on the Benefits of Water Framework Directive Programmes of Measures in England and Wales, a pre-cursor to the Environment Agency's WAG estimates for improvements in water quality. The mean WTP used within the Martin-Ortega et al. (2014) study was £49.6 per household per year. This was combined with the cost and lifetime estimates from Table 2-14 to calculate NPV for the remediating measures over three time horizons (Table 2-15). The NPV for each time horizon indicates that the benefits of improving water quality in the Thames sub-catchment significantly outweighed the costs of the measures.

Table 2-15 Aggregated costs and benefits of water quality improvement in the Thame sub-catchment (GBP)

	2015	2021	2027
Aggregated cumulative costs	19,133,243	29,172,756	36,368,221
Aggregated cumulative benefits	33,182,034	50,234,867	64,107,357
NPV	14,048,791	21,062,111	27,739,136

(Source: Martin-Ortega et al. 2014)

2.1.212 Under the same REFRESH project, Balana et al. (2014) undertook cost-effectiveness and disproportionality analyses in two Scottish sub-catchments (Loch of Skene/Leuchar Burn catchment and Tarland Burn sub-catchments), both located within the River Dee Catchment in North-East Scotland. Nitrogen in Tarland and phosphorus in Loch of Skene/Leuchar were identified as the key pressures within these sub-catchments. Table 2-16 contains the costs and lifetime of measures to achieve GES in the Tarland and Loch Skene and Leuchar sub-catchments.

Table 2-16 Costs and lifetime of measures that achieve GES in Dee sub-catchments

Measure	Investment costs (GBP in year 0)	Annual costs (GBP per year)	Lifetime (years)
Tarland sub-catchment			
10% N reduction target (3 mg N/l)			
20% reduction in applied N fertilizer	0	5,480	Continuous
Constructed field wetland (CFW) area for field interception	10,434	0	10
10 m buffer strips adjacent to arable land (with management)	0	8,580	Continuous
15% N reduction target (2.5 mg N/l)			
20% reduction in applied N fertilizer-Arable land	0	5,480	Continuous
Constructed field wetland (CFW) area for field interception	10,434	0	10
10 m buffer strips adjacent to arable land (with management)	0	8,580	Continuous
Winter cover crops (uptake of 20 kg through winter)	0	20,790	Continuous
20% reduction in applied N fertilizer-livestock (grass land)	0	35,570	Continuous
Arable grassland conversion (20% change of extensive improved grassland)	0	41,000	Continuous
Loch of Skene and Leuchar Burn sub-catchment			
20% SRP reduction target			
50% reduction of fertilizer application to grassland system	0	0	Continuous
20% reduction fertilizer application to arable land	0	1,874	Continuous
WWTW to reduce effluent SRP concentration to meet 1mg/l.	35,040	0	20

(Source: Balana et al. 2014).

2.1.213 Non-market benefits of improving water quality in the Dee sub-catchments were obtained from research by Glenk et al. (2011) in which the benefits of improving Scottish lochs and rivers ecological status to comply with the WFD were estimated at the national level, distinguishing between the two main Scottish river basins: the Scotland River Basin District and the Solway-Tweed basin. The resulting WTP estimates for the Scotland River Basin District were £1.81 per household per year for rivers and £1.2 for lochs. The aggregate non-market benefits of improving ecological status in the Dee sub-catchment is provided in Table 2-17.

Table 2-17 Aggregated non-market benefits in Dee sub-catchments.

Sub-catchment	Non-market benefits (£/year)
Tarland sub-catchment	427,567
Loch of Skene and Leuchar Burn sub-catchment	751,916
-Subtotal for Loch of Skene	*482,380*
-Subtotal for Leuchar Burn river's catchment	*269,536*

(Source: Glenk et al. 2011)

2.1.214 The aggregated non-market benefits were combined with the cost and lifetime estimates to calculate NPV for the remediating measures over three time horizons (see Table 2-18). The estimated NPVs indicate that the benefits of improving water quality in the Dee sub-catchments outweigh the costs of the measures analysed in both sub-catchments and for the three time horizons.

Table 2-18 Aggregated costs and benefits of water quality improvement in Dee sub-catchments (GBP)

Sub-catchment (Target)	Net Present Value (GBP)		
	2015	2021	2027
Tarland (10% N reduction target – 3 mg N/l)	3,265,338	4,947,214	6,318,672
Tarland (15% N reduction target – 2.5 mg N/l)	2,636,082	3,977,385	5,099,255
Loch of Skene and Leuchar Burn (20% SRP reduction target)	5,873,400	8,909,638	11,380,053

(Source: Glenk et al. 2011).

2.1.215 At a national level, Defra and the Environment Agency have identified the increments in water quality required to meet the objectives of the WFD, namely to achieve Good Water Quality status, on each length of river and area of lake, based on compliance with chemical, biological and hydromorphological conditions. Combining this information with estimates (based on Environment Agency and NERA, 2007 data) of the value of non-market benefits associated with improvements in water quality in rivers and lakes in England and Wales, Morris and Camino (2011) have estimated the aggregate value of water quality improvements across England and Wales to be £1,140 million per year.

Evidence gaps and priorities for research

2.1.216 To better appreciate how the above information fits together, we include a synthesis diagram (Figure 2-15) to represent: a) which climate change drivers are involved (1st row of diagram); b) the immediate environmental parameters that are affected by climate change drivers (2nd row of diagram); and c) the adaptations (3rd row of diagram) that are either already in place, proposed, or simply possible, that may reduce negative impacts on wildlife, human health or the economy. For this latter row, our examples focus on 'harder' adaptations of a physical nature (i.e. grip blocking, slurry tank provision) that tend to focus on a particular negative impact, rather than other, 'softer' measures (e.g. raising awareness), that tend to be targeted less specifically.

2.1.217 As is clear from this diagram, the physical aspects of how climate change can affect the water environment are relatively well resolved, with multiple sources, mechanisms and receptors identified in the top half of the diagram. However, the relative emptiness of the bottom half of the diagram suggests that, for many of the water environment adaptations that are available, evidence for their efficacy (relative to each other, or when deployed as part of a suite of measures) is more limited.

2.1.218 To facilitate future research efforts on the impacts of climate change on the water environment, we have identified a number of key research themes that we believe to be most pertinent, which if addressed would generate substantial advances in our understanding. For simplicity we break these down into a number of discrete points. These begin with more general points and finish on more specific issues.

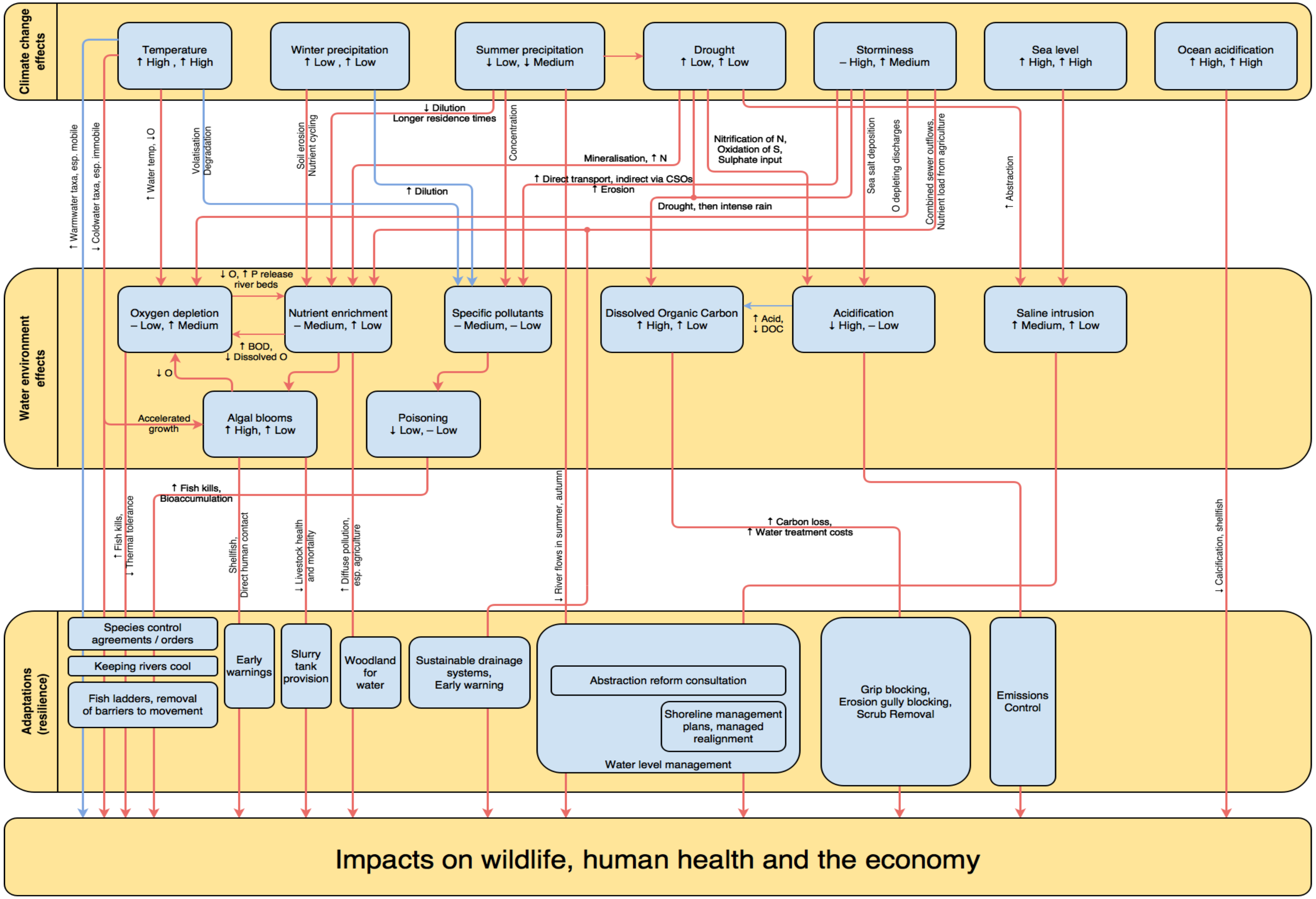


Figure 2-15 Illustration of climate change impacts on the water environment in the UK, and consequent impacts on wildlife, human health and the economy

Examples of adaptations that can dampen these impacts are also suggested. Observed and projected impacts of climate change appear in blue blocks on the first row, with information on their present day trend (↑ increasing or ↓ decreasing) and associated confidence (high, medium or low), followed by expected future trend (↑ or ↓) and confidence (high, medium or low). A dash symbol indicates no trend. Arrows link changes in climate to their impact on variables or phenomena in the water environment; red arrows indicate a negative impact, while blue arrows indicate a positive impact.

1 The importance of ‘core’ research on climate science

2.1.219 It is clear that almost all the information we have on future UK trends in climate come from either the Met Office (Hadley Centre) or the UK Climate Impacts Programme. The continued existence and funding of these institutions is essential if we are to improve our understanding of what is to come this century, and reduce uncertainties in this understanding (see also point 3 below). They also contribute a great deal of expertise and data towards international assessments of climate change coordinated by the IPCC, another key resource for scientists, policymakers and indeed politicians.

2.1.220 Despite the UK’s rich history of climate change research, much more is left to discover and understand as we move beyond basic (yet powerful) estimates of mean temperature change to improved detection of climatic extremes caused by the anthropogenic signal, and a better understanding of how extreme events (often the most damaging) may change in frequency and magnitude with climate change.

2.1.221 The lower level of confidence in future precipitation projections makes it vital that planners embrace uncertainty if they are not to promote maladaptation. The lack of certainty over even the sign of the change to be expected nationally is problematic, and it follows that there is high uncertainty at the regional level over which the water companies and authorities operate. Although there is “ongoing, albeit modest improvement” in the performance of climate models (IPCC 2013), there has been little or no convergence in the projected climates derived from models over successive model intercomparison exercises. This would suggest that improvements to model physics do not necessarily reduce the uncertainty in predictive outcomes (Murphy et al. 2009). Furthermore, all models contain some systematic biases – such as the inability to simulate blocking episodes accurately- which cannot be seen from the results of intercomparison projects. Thus decision makers cannot necessarily wait for the climate models to provide certainty, and rather, must act without complete information.

2 Better data

2.1.222 ‘Water quality’ is a simple term that belies huge complexities in the water environment, and the continuing debate over how it should be measured serves to highlight how many different ways there are of achieving this. In 2015, we are at the end of the first cycle of WFD planning. As the means of monitoring quality under this framework are now set, it is important that the same variables are now measured, consistently, over a number of years. Similar to the Environment Agency’s General Quality Assessment programme, this would allow any trends in quality to be established. Responses to short-term (interannual) climatic variation could also be discerned. This will allow scientists (in the water authorities, and in academia) to identify where problem catchments are, and the sorts of quality measures that are more likely to present a problem for WFD compliance as the climate shifts. A focus on data sharing would also allow this analysis ‘workload’ to be borne by a wider community of researchers beyond the water authorities (including NGOs, academics, and research agencies).

2.1.223 Better data also applies to the wider environment: for example, spatially explicit modelling of peatland degradation is often complicated by the existence of numerous non-linear and threshold effects (Glenk et al. 2014), yet there is no national (UK wide) map of degradation. Better data on peat depth and extent using a consistent classification framework is also required, as in some cases the best data we do have (e.g. 1980s Lowland Peat Survey) are increasingly out of date. Funding for habitat mapping (including land use/land cover) should unquestionably continue, and the new peat map for Wales (based on BGS and NRW data, Evans et al. 2015) illustrates the wider benefits of good habitat data (which were used in its construction). Owing to how critical the impacts of land management are for many of the water quality issues in the uplands, a similar map documenting land management would prove invaluable. Risk maps for other pressures on the water environment (e.g. eutrophication) would also help target mitigation measures, or direct further, more focussed research approaches.

3 The uncertainty cascade

2.1.224 The UK climatological research community is leading the world in 'grasping the nettle' of handling uncertainty, and in 2009 produced the world's first set of probabilistic climate change scenarios (UKCP09). However, some six years after the publication of UKCP09, only a small minority of climate and water scientific publications have utilised these scenarios, and many still rely on older scenario sets (Arnell et al. 2015). Uncertainty- relating to the choice of climate model and emissions scenario- cascades down the impacts chain to the water variables measured by scientists and under the WFD. This uncertainty should at the very least be acknowledged when communicating the results of studies investigating the impacts of climate change, yet it is often ignored. Basic sensitivity analyses designed to test underlying assumptions are rare.

2.1.225 There is a clear need for the research community to find a means of including uncertainty over climate projections into their approach. Given that the preference of many scientists would be for (often expensive) strategic decisions to be based on the evidence, a greater emphasis should be placed on evidencing these decisions properly. If analyses of impacts take a full account of uncertainty, the true range of projected changes can be established, and decision makers can formulate suitable adaptations that offer benefits across this range.

4 Geographic spread of research effort

2.1.226 It remains the case that a large proportion of UK water research has been conducted over a relatively small subset of UK catchments, management contexts, and habitat types. Northern Ireland has largely been neglected. Despite improvements in spatial coverage, it is important that the UK ECN, UWMN and UK peatland networks be expanded into more catchments of concern and into sites representing the full diversity of UK habitats, because different types of catchment respond differently to the same environmental changes. It is also as important to continue and expand lowland catchment monitoring programmes such as the LWEC Demonstration Test Catchments in Cumbria, Hampshire and Norfolk.

5 Adaptation experiments

2.1.227 The evidence to link climate change with its impacts on water quality parameters is growing, but it remains the case that for many of the adaptations designed to alleviate these impacts, we simply do not know how effective they might be. Similar to the principles of evidence-based policy (Cabinet Office 2012) or evidence-based conservation (POST 2011), establishing the efficacy of adaptations should be a priority if we are to invest a substantial amount of capital (political and economic) in adapting our natural systems to climate change. There are notable interventions for which research has generated specific guidance for landowners or policymakers (e.g. riverine shading), and these projects demonstrate the value of experiments in adaptation. But even here, monitoring must continue to establish the long-term consequences of these interventions.

2.1.228 To facilitate our understanding of adaptation efficacy, we must also understand how patterns of climate change will manifest at all spatial scales. To do this will require an expansion of temperature monitoring effort, because although the UK has an efficient and reliable meteorological station network on land, the patterns of change in aquatic environments are less well described. In a recent study of the data in the Environment Agency's Surface Water Temperature Archive (for England and Wales), water temperature was found to have increased by 0.03°C per annum between 1990 and 2006 (Des Clers et al. 2008, confirmed by Orr et al. 2015). However, these important studies remain the only national estimates we have (Watts et al. 2013). In their review of the literature on UK river temperature, Hannah and Garner (2015) concluded "Most UK river temperature investigations have been restricted to the (sub-)basin scale and by relatively short observational records. Consequently, there is limited knowledge of spatial and temporal variability of river temperature at the inter-basin to regional scale".

2.1.229 Thus it is unclear how coupled the water temperature of aquatic environments will be to broader-scale warming, and many assessments of the implications of climate change for the water environment (such as this review) rely on the assumption that broad-scale patterns will be replicated at ground level. The validity of this assumption could be tested via intensified monitoring of water temperature in the field, particularly in areas (e.g. the uplands) that are underrepresented by current monitoring regimes (Orr et al. 2015). Linking this monitoring to studies of biological change will also be required.

6 Adaptation Monitoring case study: The 'Keeping Rivers Cool' project

2.1.230 The Environment Agency project 'Keeping Rivers Cool' is an ongoing (2012-2016) adaptation project specifically targeting riverine species at risk from warming temperatures. Broadly, the aim is to increase the provision of riparian shade by planting trees, erecting fences and promoting natural regeneration. This will increase the potential for cooler microclimates to form, which under climate change will more closely match the pre-warming thermal niches of native species (Suggitt et al. 2011). For example, only 20-40% shading along a river stretch in the New Forest was enough to keep river temperatures below the lethal limit for Brown trout (*Salmo trutta*) in the New Forest (Broadmeadow et al. 2011).

2.1.231 The target demonstration areas for 'Keeping Rivers Cool' are headwaters for salmon and trout rivers, such as the River Ribble, the Hampshire Avon and the Wye. The learning from these demonstrations has been captured in a detailed guidance document, including recommendations on what to plant and where, based on the orientation of the stream, its situation in the catchment and the target species (Lenane 2012). A recent study confirmed that the approach is successful at reducing stream temperature in a variety of regional (European) climatic contexts (Verdonschot et al. 2014). Should it be applied more widely, this approach would form part of a suite of possible measures to 'resist' or 'offset' warming for native species at risk from its impacts (Greenwood et al. 2015).

Conclusions

2.1.232 Climate change has had a multitude of impacts on the water environment, and is expected to have many more as temperatures continue to rise this century (Watts et al. 2013). Here, we have reviewed those impacts of greatest concern, in as systematic a manner as possible. The focus was on establishing the present and future trends in these impacts, identifying the influence of climate change, and discussing potential adaptation measures. We also provided the latest information on the state of climate change science, and the condition of the water environment: both are important starting points for any assessment of the risk climate change poses to any natural asset. We provide a brief summary of the key risks for each country below, although we would emphasise that many of the challenges each country faces are shared:

- England: Over-abstraction, nutrient enrichment, increased frequency of CSO discharges, and the continued release of DOC from upland peat.
- Scotland: Nutrient enrichment, increased frequency of CSO discharges and DOC release.
- Wales: Over-abstraction in the east, nutrient enrichment, DOC release in the uplands.
- Northern Ireland: Nutrient enrichment, increased frequency of CSO discharges.

2.1.233 Although we referred to hundreds of scientific papers, reports and policy documents, there is a wealth of studies that are not included here due to project constraints. However, we would argue that even in this broader literature, there is: 1) an emphasis on establishing physical processes between climate and water variables, but 2) there is little emphasis on establishing how land management or other human impacts (and adaptations) interact with climate and water to mitigate (or indeed exacerbate) adverse impacts. Addressing such research questions is by definition a multi-disciplinary endeavour and one that will require a large amount of coordination and partnership working to achieve. Early results from the new RCUK Doctoral Training Centres (or Centres for Doctoral Training) illustrate the substantial gains that such an approach brings.

2.1.234 The time it will take to implement adaptation actions at a sufficient scale, and for the water environment to respond, means that adaptive management needs to be taken at a scale that is matched to the challenge. Although the need to implement this management is pressing, we can afford to await the results of pilots. These pilots will need to test for the relative influence of multiple pressures on the water environment, but also the relative merits of multiple adaptations designed to alleviate them. A well-designed, integrated approach will be required.

2.1.235 There is a need to focus on land-based actions that result in win:wins. Landscape solutions should be explored that seek to mitigate several pressures simultaneously, and hence provide multiple benefits. For example, targeted woodland planting can simultaneously: a) mitigate flood risk, b) provide increased shade and a cooler microclimate for watercourses, and c) buffer the transport of nutrients from agriculture into those watercourses. Strategic decisions will need to address the full range of likely variation in projected changes and their impacts, and this range will be larger for some variables (and seasons) than others. Adaptation measures are required that will be beneficial whatever the extent, rate or even direction of climate change.

2.1.236 There is also a need for more quantitative evidence relating to the costs and benefits of the impacts of climate change on water quality and the implications of this on water treatment costs, human health, opportunities for recreation, amenity and biodiversity. While there is an increasingly sophisticated array of valuation techniques that can help us to derive values, we are limited by our understanding of the relationships between climate change (as well as other drivers), ecosystem response and the consequent effects on human well-being. Targeted research supported by more extensive monitoring (e.g. of the impacts of CSO overflows on human health and the effects of low flows on water quality) will help better define these exposure-response relationships.

2.1.237 We conclude by offering the following overarching themes to our review:

1. Our understanding of the impacts of climate change is better for certain water body types than it is for others. In relative terms, our understanding is better for rivers and lakes than it is for groundwater, estuarine, transitional and marine areas.

2. There is more uncertainty over what to expect from some aspects of climate change than for others. For example, it is virtually certain than mean temperature will increase everywhere in the UK, whereas even the direction of precipitation changes (at some times of year) is uncertain.

3. Direct human impacts exert a huge influence on water quality (especially for nutrient status and abstraction), yet little is known of how they vary even in present-day conditions. Lack of data is holding back our understanding in this area.

4. Abstraction pressures, the intensity of land management and climate change are all acting to render water systems less able to cope with stressors (such as low flows) which it would otherwise be resilient to.

5. Evidence for the relative efficacy of most adaptations targeting aquatic systems is equivocal. The conditions in which these different adaptations may be more (or less) appropriate are also unclear.

6. A substantial amount of research has been completed, but there is still much to be done. The complexity of the water environment and the number of potential modifiers involved probably mean that it is unlikely to ever be conclusively understood in its entirety. Prioritising where this research is needed most is therefore important.

Equable climate

3 Equable Climate

Introduction

3.1.1 Here, we take ‘equable climate’ to be a public good delivered by the climate regulation ecosystem service (National Ecosystem Assessment, NEA 2011). In the context of anthropogenic climate change, assessing the risk to an ‘equitable climate’ relies on identifying and quantifying the most important sources, stocks and sinks of greenhouse gases, particularly carbon dioxide. Providing this information as annual submissions is vital for the UNFCCC (United Nations Framework Convention on Climate Change) process; yet even for industrialised (Annex I) countries, data on the reservoirs of carbon can be inadequate. We begin with a review of the latest carbon stock estimates (Section 0), and document what evidence exists on potential changes in these stocks (Section 0).

3.1.2 To ensure the UK’s contribution towards an equable climate, potential risks to carbon stocks must be identified and minimised. In the quantitative components of this chapter, we focus on the potential loss or gain of carbon to/from the atmosphere. We take ‘loss’ to be onward transport of carbon (e.g. from the catchment, the landscape, the forest, etc), whilst acknowledging that ‘onward’ does not always mean that the carbon is lost directly to the atmosphere. We discuss some of these other ‘onward’ effects (e.g. Dissolved Organic Carbon export into watercourses) in the clean water component of the Project C assessment. We also emphasise that the local or regional impacts of carbon loss (e.g. Dissolved Organic Carbon export into watercourses) can be as detrimental to those environments as the realised (or potential) impacts of climate change.

3.1.3 In this chapter, two key drivers of change in carbon stock are examined: land cover change, and the indirect effect of climate change on land cover. Patterns of land cover are of critical importance to carbon stocks; while this importance is obvious in terms of the vegetation present above the surface, the effects are also evident in the carbon content of the soil below the surface. We therefore assess the effect of land cover changes to carbon stocks in both: a) the soil, and b) the vegetation. We use a land cover dataset that also accounts for the indirect effect of climate change. This allows us to quantify how warming effects, such as increased droughting or the abandonment of agricultural land, drive changes in land cover and thus changes in carbon stock. Our analyses also capture the effect of adaptation options designed to reduce these negative effects (or take advantage of positive effects).

3.1.4 Overall, our analyses provide an indicative idea of the final, equilibrium change in carbon stocks associated with a number of future scenarios of land cover. We include estimates of the value of changes in carbon stored in soil and vegetation stocks under a range of climate scenarios (Section 0). We also include a brief discussion of how further research on this topic could proceed (Section 0) and finish by offering some conclusions (Section 0).

Quantifying present day carbon stocks in the UK

3.1.5 This chapter estimates the carbon stock of two important reservoirs of terrestrial carbon: soil, and vegetation. In each section, we discuss the development of datasets designed to quantify these stocks, before providing justification for the data used to assess risks to these stocks.

Soil carbon stocks

3.1.6 Soils are the largest reservoir in the terrestrial carbon cycle (Petrokofsky et al. 2012). Peatlands represent one of the most dense stores of terrestrial carbon in the world, containing 500Gt of the 2200Gt of the world’s soil carbon store on only 4 million km2 (= 3%) of its area (Victoria et al. 2012). The UK is among the top ten nations in the world in terms of its total peatland area (CCC ASC 2013), and due to focussed survey effort, has some of the best data documenting the characteristics of these landscapes.

Existing soil carbon stock estimates

3.1.7 Historically, surveys of the soil were carried-out for agricultural purposes, with surveys in peatland conducted with a view to quantifying its potential for extraction. National surveys were conducted in the 1980s (National Soils Inventory or NSI in England and Wales, Lowland Peat Survey, Soil Survey of Scotland, Department for Agriculture and Rural Development soil survey in Northern Ireland), with inventories compiled in the 1990s. These initial inventories faced some difficulties over the extent to which the full depth of soils (and peats in particular) were represented, with constituent countries of the UK reporting carbon stock to different depths. The country-level inventories also used different classification systems for soil type.

3.1.8 For the purposes of UNFCCC reporting, it was recognised that quantification of soil carbon present at certain depths (0-30cm, 30-100cm) needed to be standardised across the UK, and a new UK-wide inventory was thus compiled (Bradley et al. 2005). The National Soil Resources Institute (NSRI, formerly Soil Survey & Land Research Centre) harmonised the data between the different inventories and produced the first national soil map (Figure 3-1) to standard peat depths at a fine horizontal grid resolution (1km x 1km). The map facilitated several improvements in our understanding of the important land cover types for soil carbon stocks. Since then, improved estimates have been derived in Scotland (Ecosse 2007), England (Natural England 2010) and Wales (Jones and Emmett 2013) and in the UK's forests (BioSoil: Vanguelova et al. 2013). We tabulate these estimates below (Table 3-1). These revised estimates use methods of converting bulk density to fractional carbon content (Countryside Survey 2010, Ecosse 2007) and better data on peat depth (Ecosse 2007, Natural England 2010). However, existing estimates still assume carbon to be distributed continuously throughout the soil profile, which is rarely the case. Therefore, there is also likely to be an increased role for 3D modelling of soil organic carbon, once the uncertainties are reduced (Poggio and Gimona 2014).

The Bradley et al. (2005) soil carbon layer

3.1.9 The Soil Survey data collated by Bradley *et al.* (2005) represent the last 'best guess' soil map for the UK, in which a standardised approach was used, assumptions were harmonised, and a consistent method adopted. The national soil database that was generated in this paper still forms the basis for UK NIR submissions for the LULUCF sector (Table 3-1). We therefore use the Bradley et al. (2005) data (provided to the project team by NSRI) for soil carbon analyses. Throughout these analyses, we used the soil carbon layer of 0 – 100cm depth (Figure 3-1 c), as estimates for carbon below 100cm in Bradley *et al.* (2005) were only available for Scotland.

3.1.10 For illustration purposes, we also show the differences between Bradley et al. (2005) and more recent estimates Figure 3-2, Figure 3-3). While estimates in upland peats are relatively similar, the estimates in lowland areas differ. The differences are most pronounced in England, where improved information on peat depth (for below 1m depth) suggests that the carbon stored in fen, reedbed and raised bog soils was previously underestimated (Natural England 2010). The authors do point out, however, that much of this lowland peat may have already been lost to cultivation and wastage, and almost all of it is subject to ammonia pollution (Natural England 2010). In the Fenlands of East Anglia, soils are estimated to be wasting at 2.1 cm/year, emitting 0.4 TgC/yr (Holman 2009), equivalent to 9% of the annual loss rate reported by Bellamy et al. (2005).

3.1.11 Given uncertainty in carbon stock estimates (Table 3-1), and the global significance of the UK's soil carbon stocks, an update of Bradley et al. (2005) could be a focus of future research.

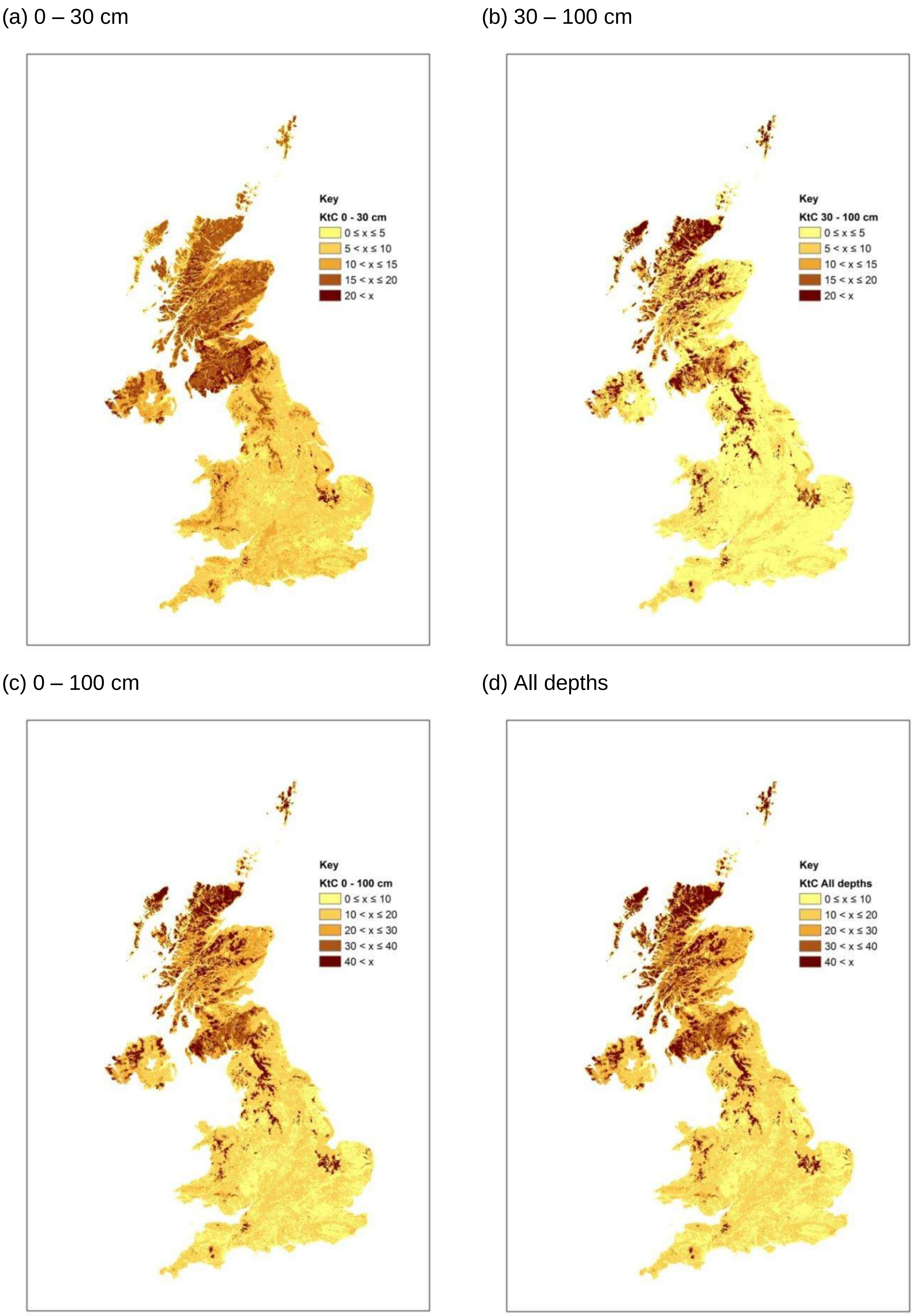


Figure 3-1 Bradley et al. (2005) estimates of soil carbon

(At: (a) 0 – 30 cm; (b) 30 – 100 cm; (c) 0 – 100 cm; and (d) All depths. Grid cells 1km width.)

Table 3-1 Collated recent estimates of the UK's soil carbon stock

Country	Depth											
	0 to 15cm	0 to 30cm	0 to 100cm					Below 100cm			All depths	
	Countryside Survey (2010)	Bradley (2005)	Bradley (2005)	Ecosse (2007)	Chapman (2009, Peatland only)	Jones and Emmett (2013)	UK NIR FCCC (2014)	Milne (2001)	Bradley (2005)	Chapman (2009, Peatland only)	Ecosse (2007)	Natural England (2010, Peatland only)
England	795	1,015	1,740	–	–	–	1,740	–	-–	–	–	584
Scotland	628	1,161	2,187	2,249	1104 ± 44	–	2,768	3,248	483	516 ± 55	485	–
Wales	159	194	340	144	–	436 ± 27	340	–	–	–	52	–
Northern Ireland	–	172	296	–	–	–	296	–	–	–	–	–
Nationwide	–	2,543	4,563	–	–	–	5,144	–	–	-	–	–

(Units Mt = Tg, see also Figure 3.1). Dashes indicate that the stated report or authors did not produce an estimate for the country in question.)

(a) Ecosse (2007, all depths) **(**b) Bradley et al. (2005, all depths) (c) Natural England Peatlands (2010, all depths)

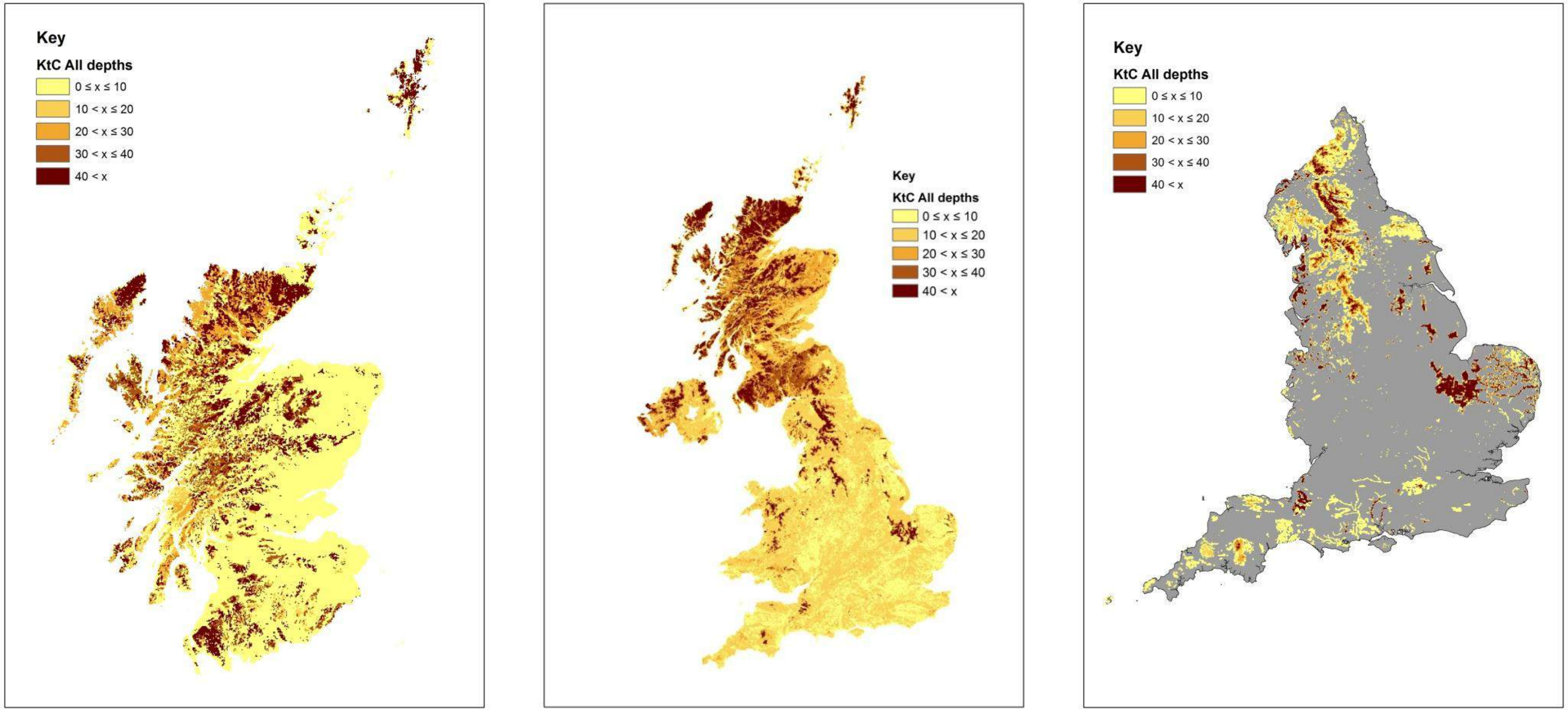


Figure 3-2 Existing estimates of soil carbon stock (units KtC, grid cells 1km width)

Panel (a) is derived from Ecosse (2007, all depths); (b) is derived from Bradley et al. (2005, all depths); and (c) is derived from Natural England (2010, all depths, peatlands only).

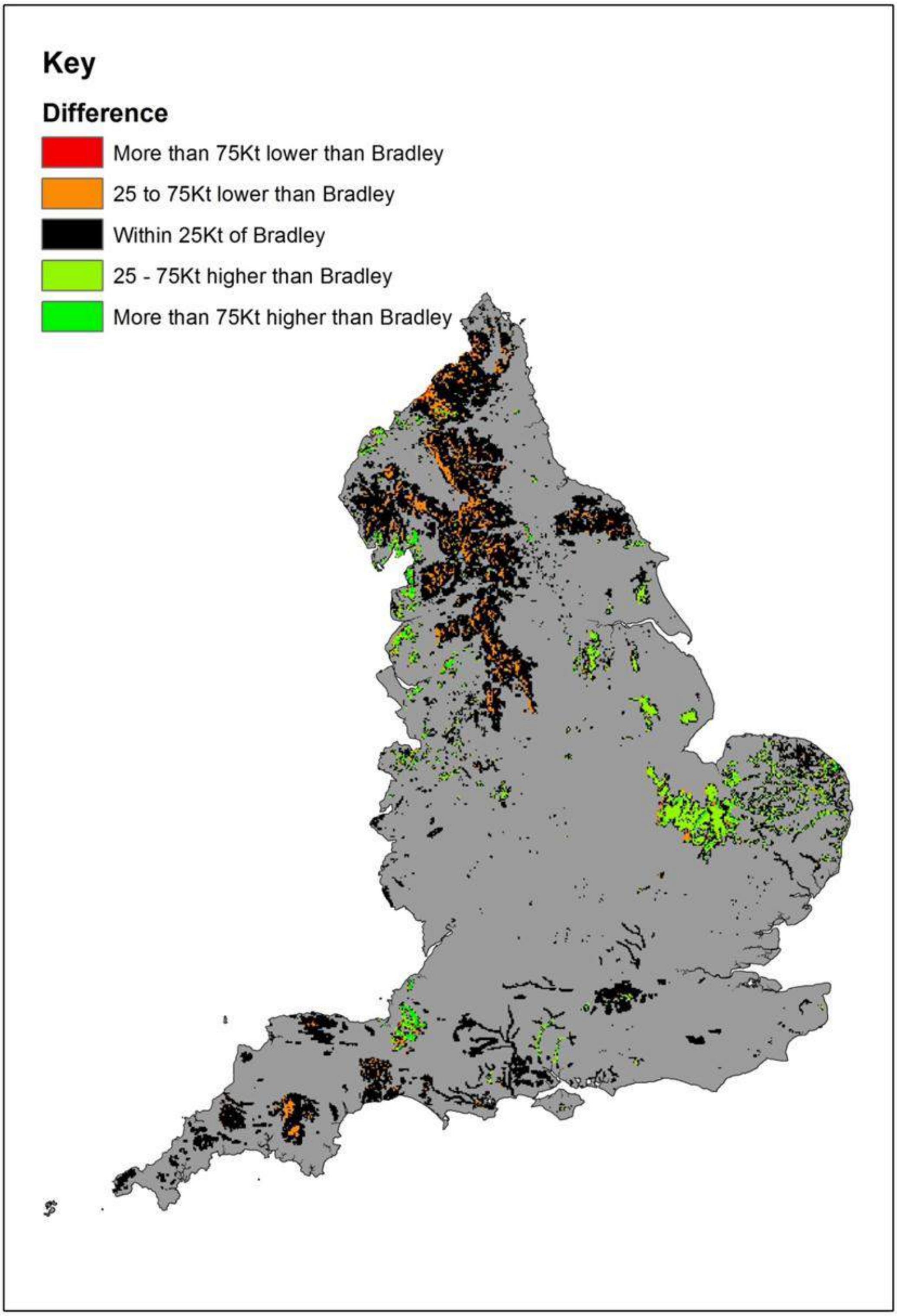


Figure 3-3 The difference between two major estimates of the carbon stocks in England's peatlands

The map shows the difference in carbon stock (KtC, gridcells 1km width) between estimates derived by 1) Natural England (2010), and 2) Bradley et al. (2005). Values represent values in 1) subtracting the values in 2). Thus green colours show a positive difference (areas where estimated KtC was higher in Natural England 2010), while red colours show a negative difference (areas where estimated KtC was lower in Natural England 2010).

Vegetation carbon stocks

3.1.13 The most recent 'bottom up' estimate of woodland carbon stocks was published by the Forestry Commission using data from the first five year cycle of the National Forest Inventory (Table 3-2). These data were used to "estimate of the amount of carbon in living trees within British woodlands" (as at 31st March 2011), including "roots, stems, branches and leaves" but excluding "biomass contained in other vegetation associated with the stand (e.g. shrubs and herbs)" (Forestry Commission 2014a). Broadleaf trees were assumed to be in leaf. The headline estimate for woodland carbon stock was 213 Mt for Britain (England, Wales and Scotland).

Table 3-2 Collated estimates of the UK's vegetation carbon stock.

Authors (Year)	Scope	Geographic area	Estimate (MtC)
Milne and Brown (1997)	All vegetation	Great Britain	117
Broadmeadow and Matthews (2003)	Forest biomass	UK	150
Forest Research (2012)	Woodland standing biomass	UK	162
Forestry Commission (2014a)	Live trees in woodlands	Great Britain	213

3.1.14 Estimates of the carbon content of woodland have tended to increase as previous studies were found to have underestimated the carbon held in root stock. Often, half the dryweight of the vegetation biomass is assumed to be carbon, but only half the carbon from the harvested tree is in the stemwood (McKay et al. 2003). So estimates derived from the weight of the harvested trees would also underestimate the vegetation carbon stock. More recently, the 'jump' in the Forestry Commission/Forest Research woodland carbon estimate from 162Mt to 213Mt is due to the move from the National Inventory of Woodlands and trees (NIWT) to the National Forest Inventory (NFI), with the more sophisticated NFI raising the estimates of both the forest area and the number of trees as a result of improved technology (such as GIS, and remote sensing).

3.1.15 Until recently, the UK NIR used C-FLOW to estimate the carbon stock in forests, but it has now switched to CARBINE, which places more emphasis on production (Matthews and Broadmeadow 2009). For non-forest habitat types, Milne and Brown (1997) is used.

Observed change in carbon stocks

Soil carbon

3.1.16 Data from the repeated surveys of the National Soil Inventory Monitoring Programme (1978 and 2003, Bellamy et al. 2005) indicate that topsoils in England and Wales (0 – 15cm depth) are losing carbon at a rate of 0.6% (4 million tonnes) per year. However, estimates derived from repeat surveys of Countryside Survey locations (1978 and 2007, Carey et al. 2008), showed no change in topsoil carbon over this period. A Defra research project specifically examined the discrepancy between the findings derived from the NSI data and those of the Countryside Survey (Kirk et al. 2011). This project tested and ruled out several plausible reasons for the discrepancy, concluding that the "the reasons for the difference between the two surveys are not yet clear".

3.1.17 Various studies have sought to explain the soil carbon loss reported by Bellamy et al. (2005), and there is continuing debate as to its cause(s). The authors themselves concluded that the rate of carbon loss was not related to land use change, and suggested that climate change was the cause. However, a study using vegetation modelling demonstrated that warming could only have contributed to 10-20% of the reported loss (Smith et al. 2007). In a follow up, Kirk and Bellamy (2010) applied a similar method to the soil data and confirmed that climate change could only be responsible for 10% of the loss they reported, concluding that continued changes in land use and management were responsible, and noting that this pattern was also evident internationally.

3.1.18 Scottish repeat surveys (over a 19–31-year period, mean 25 years) showed no net change in total Scottish soil carbon stock (Chapman et al. 2013). This corroborated Countryside Survey findings for Scotland which suggested no change in Scottish topsoil between 1978 and 2007 (Carey et al. 2008). The total soil carbon stock in Northern Ireland has also shown no sign of a trend, although stocks are threatened by urbanisation (Tomlinson and Milne 2006).

3.1.19 UK NIR (2014) used a dynamic model of soil carbon stock change to calculate Land Use, Land Use Change and Forestry (LULUCF) contributions to the national emissions inventory. The LULUCF sector is the only sector in the inventory to report a net carbon sink (since 1998), with net emissions of – 5 Mt CO_2 per annum since 2004. Soil carbon sequestration and gains in above and below ground biomass are cited as drivers of this trend.

Vegetation carbon

3.1.20 Changes in vegetation carbon can, to a degree, be inferred from changes in forest cover. Estimates indicate that woodland cover was down to 5% of the country's land area by the end of the 19th century, due to historic, long-term deforestation (Smith 2001). However, since then, forest cover has generally increased, with recent increases in the 0.25-0.5% per annum. Woodland cover in Britain currently (as in 2011) stands at 13% (Forestry Commission 2013).

3.1.21 The constituent countries of the UK have a number of strategic targets to increase forest cover further (Table 3-3). These expansions represent an important opportunity for the UK to sequester carbon. Note that the target for Wales was aspirational.

Table 3-3 Forest cover estimates of UK countries, and targets for increasing them

Country	Cover estimate	Target
England	10% (FC 2013)	12% by 2060
Scotland	18% (FC 2013)	25% by 2050
Wales	15% (FC 2013)	Increase of 4.8% by 2030 (aspirational target)
Northern Ireland	6% (Forest Service 2006)	12% by 2055

3.1.22 The linkage between new planting and carbon uptake is complex. Peak uptake of CO_2 occurs once the stand is established and the canopy is closed, while uptake declines in older trees (Forest Research 2012). Furthermore, the management techniques required to maximise CO_2 uptake (i.e. maximise productivity) and maximise total carbon stock differ (Forest Research 2012). Thus the phasing of new planting (and replacement of existing stock) and priorities for continuing forest management will affect annual emissions reporting in the NIR, and possibly progress towards climate mitigation goals.

Changes in soil carbon stocks arising from land cover change

Justification

3.1.23 The NEA concluded that "future changes in land use could have as much impact on ecosystem services as the direct effects of climate change" (NEA 2011). It is therefore critical that the implications of changes in land use/cover be taken into account when assessing carbon stocks.

Yet quantifying the implications of land cover change is notoriously difficult. Produced in 1990, CEH's Land Cover Map (LCM) was the first satellite-derived estimate of land cover; this map, and its updates (2001, 2007), have facilitated numerous analyses of the effects of land cover at the national scale. In the interim, other means of habitat classification have been produced to take advantage of remote sensing data, but few have benefitted from the level of ground-truthing and post-processing that these national maps were subject to.

3.1.24 The CEH LCM data formed the basis for the development of a scenario set for future land cover under the National Ecosystem Assessment (NEA 2011). Any future scenarios of land cover will necessarily be speculative, but the NEA dataset represent an important means of testing the implications of these possible changes for UK carbon stocks, and thus we adopted the data within our assessment. In this section, we assess the level of possible changes to soil carbon stocks arising from these land cover changes, while in the following section we assess the issue in terms of its effects on vegetation carbon stocks.

Data

The National Ecosystem Assessment land cover scenarios

3.1.25 The National Ecosystem Assessment (NEA 2011) provided estimates of the land cover in the year 2060 based on various socio-economic scenarios (the 'NEA scenarios', n=6) and under 'high' and 'low' climate change. All the NEA scenarios assume a global decline in resource availability, but differ in terms of their assumptions about the public's environmental awareness, wellbeing, ecological footprint, and the levels of governance and adaptation capacity that would be realised over the next 50 years. We selected three of these scenarios for further analysis: 'Local stewardship', 'World Markets' and 'Green and pleasant land' (Table 3-4).

Table 3-4 The properties of the three NEA scenarios analysed in this chapter.

	NEA scenario		
	'Local stewardship'	'Green and pleasant land'	'World Markets'
Summary	"This is a future where society is more concerned with the immediate surroundings and strives to maintain a sustainable focus on life within that area".	"A preservationist attitude arises because the UK can afford to look after its own backyard without diminishing the ever-increasing standards of living".	"High economic growth with a greater focus on removing barriers to trade is the fundamental characteristic of this scenario".
Indirect effect of climate warming	Reduces arable but increases native wood planting (not beech or other climate change intolerant species). Some improved grassland is converted to semi-natural grassland because it is more climate change tolerant.	Higher temperatures will affect some land cover types—arable suffers a slight loss with little adaptation capacity (semi-natural grassland gains here). Broadleaf woods also suffer slightly as beech and some oak woods cannot cope with climate change in southern UK.	Very little adaptation capacity in WM; High climate change reduces arable area in south (abandoned to semi-natural grassland or southern hemisphere conifers). Some broadleaf woods suffer and are converted to conifer.

(Source: Adapted from NEA 2011.)

3.1.26 Each NEA scenario came with two climate change versions: 'low' and 'high'. These represent two possibilities given the level of climate change experienced (See Table 3-5). The low and high versions are loosely based on the headline results for mean temperature and precipitation changes under the UKCP09 low (SRES B1) and high (SRES A1FI) emissions scenarios for

2050–2079, although it must be highlighted that the UKCP09 scenarios were not applied by NEA in a spatially explicit manner. These low and high scenarios are projected to drive changes in global mean temperature of +1.8°C (likely range +1.1 to +2.9°C) and +4.0°C (likely range +2.4 to +6.4°C) respectively (IPCC 2007).

3.1.27 In the NEA analysis (NEA 2011), the low and high climate versions of the scenarios could drive differing approaches to land management. This worked on the premise that a higher level of temperature change is more likely to force a change in land cover than a lower level of change. Hence, for example, conversions to more climate-tolerant habitats (such as semi-natural grassland) are more likely under high climate versions; similarly, more areas of arable farming are also abandoned under the high climate change, as the land becomes uneconomic to farm. However, the effect of climate on the vegetation itself was not modelled mechanistically, i.e., effects on vegetative succession or growth were not considered. Thus, an indirect effect of climate change is included, e.g. high temperature change affecting agricultural land use, but not a direct effect of climate change on physical processes, e.g. accounting for interactions between the vegetation and the atmosphere in the model.

3.1.28 The impact of the low and high climate versions was also reflected in the different levels of adaptation commitment in the NEA scenarios, as scenarios with a higher level of commitment experience fewer negative effects of high climate change. So for example, under 'World markets' (low adaptation commitment), the difference between 'low' and 'high' climate versions is quite large (in terms of the difference in percentage cover of the broad habitats), as what adaptations policy is largely reactive, and focussed on high value assets only. The other two scenarios ('Green and pleasant land', 'Local stewardship') have a higher level of commitment to adaptation, and so for these scenarios the difference between the 'low' and 'high' climate versions is generally less.

3.1.29 We provide a table documenting the changes in land cover estimated by NEA (2011) below (Table 3-5).

Table 3-5 Percentage change in land cover by scenario and low/high climate version

NEA broad habitat	NEA scenario and climate version					
	Local stewardship		Green and pleasant land		World markets	
	High	Low	High	Low	High	Low
Arable	– 5.1	– 3.9	– 7.7	– 6.4	+ 2.4	+ 4.2
Improved grassland	– 3.1	– 1.8	– 6.9	– 5.9	– 6.6	– 6.4
Broadleaved woodland	+ 1.4	+ 0.4	+ 4.8	+ 5.7	– 1.1	– 0.6
Conifer	– 0.6	– 0.6	– 1.5	– 1.6	+ 0.9	– 0.3
Urban	– 0.4	– 0.2	within ± 0.05	within ± 0.05	+ 8.1	+ 8.3
Semi-natural grassland	+ 6.3	+ 5.9	+ 9.7	+ 6.4	– 2.3	– 2.8
Upland	+ 0.4	+ 0.2	+ 0.8	+ 1.0	– 2.5	– 2.7
Water	+ 1.1	within ± 0.05	+ 0.8	+ 0.7	+ 0.9	within ± 0.05
Coast	within ± 0.05	within ± 0.05	within ± 0.05	within ± 0.05	– 0.1	within ± 0.05
Sea	within ± 0.05	within ± 0.05	within ± 0.05	within ± 0.05	+ 0.3	+ 0.3

(Source NEA 2011).

Methods

3.1.30 Bradley *et al.* (2005) provided estimates of the carbon stock of 'land types' (given in Table 6 of Bradley et al. 2005, repeated in Table 3-6). We used these values to calculate the amount of carbon in each 1km grid cell under each NEA scenario (n=3, 'Local stewardship', 'Green and pleasant land', 'World Markets') and climate (n=2, 'high' and 'low') combination. The Bradley *et al.* (2005) 'land types' do not map onto the NEA 'broad habitats' precisely, so we used the scheme set out in Table 3-7 to translate the Bradley *et al.* (2005) values into suitable values for the NEA broad habitats. Similar to UK NIR (2014), we calculated "non-spatially-explicit" conversions between land cover types, and this approach is used widely in greenhouse gas accounting.

Table 3-6 Soil carbon densities reported for land types

Bradley et al. (2005) land type	Soil carbon density (kg / m^2) to 1m depth
Semi-natural	32
Woodland	25
Pasture	16
Arable	12

(Source: Bradley et al. 2005.)

Table 3-7 Scheme to match the Bradley land types to the NEA broad habitats.

Bradley land type	NEA broad habitat
Semi-natural	Mountains, Moors, Heaths
	Semi-natural grasslands
Woodland	Woodlands
Pasture	Improved grassland
Arable	Arable

(Source: Bradley et al. 2005 and NEA 2011.)

3.1.31 The estimated changes represent the final, equilibrium changes in carbon content following the change in land cover, and thus they can take anything from 50-750 years to complete (UKNIR 2014). Thus they must not be taken as indicative of the carbon content of grid cells in the year 2060; rather, they are indicative of the long-term (> 50 year) risks and opportunities facing soil carbon stocks under land cover change.

Sensitivity analysis based on soil depth

3.1.32 The opportunities for carbon sequestration on shallower soils may be limited, or indeed non-existent. We therefore calculated estimates of changes in stock after screening out 'shallow' and 'intermediate-shallow' grid cells contained within the British Geological Survey Soil Thickness map (resolution 1km grid cell width). The BGS (2015) note that: "As a rule of thumb, these categories are based on the ability to manually dig (with a spade) without being hindered by a substrate that is too strong to excavate (i.e. layers of solid rock, dense gravel, or very stiff clays). An example of a deep soil would be one developed over unconsolidated, clayey Quaternary deposits, and an example of a shallow soil would be one found in areas underlain by chalk or limestone". We enclose the category descriptions for this map (Table 3-8).

Table 3-8 Key to the British Geological Survey soil thickness map

Soil Thickness	Definition	Included in estimates derived from this sensitivity analysis?
Deep	A thick soil profile is likely. Soil (and any underlying parent Material) should be easily dug to a depth of more than 1m.	Yes
Deep-intermediate	The soil profile may vary from thick to intermediate. Soil (and any underlying Parent Material) can be dug to a depth of 1m and possibly more in some places.	
Intermediate	A 'typical' soil profile is likely. Soil (and any underlying Parent Material) can be dug to a depth of 1m.	
Intermediate-shallow	The soil profile may vary from thin to intermediate. The underlying Parent Material is potentially difficult to dig at depths greater than 0.5m.	No
Shallow	A thin soil profile is likely. Digging the Parent Material beneath the soil will be extremely difficult at a depth of 0.5m (or possibly less).	
Not applicable	No data applicable (typically inland water bodies).	

(Source: British Geological Survey).

3.1.33 Thus only cells where the soil can be dug to a depth of 1m were included in this sensitivity analysis. The removal of these cells represented in a reduction of 59,923 km^2 in the land area subject to analysis.

Results

3.1.34 Our analysis suggests that the relative implications for soil carbon under the three scenarios are quite different (Table 3-9, Figure 3-4). 'Local stewardship' and 'Green and pleasant land' appear similar in policy terms (NEA 2011, p. 1253), and both result in substantial gains in soil carbon throughout lowland Britain. However, the areas of highest gains identified under each scenario are substantially different. In 'Local stewardship', the highest gains come in upland periphery areas, where afforestation and changes from improved grassland to semi-natural grassland drive a long-term increase in carbon amounts. Large parts of lowland England fall within the range +1 to +3 ktC per km^2, while most of the gains in upland periphery areas range from +3 to +5 ktC per km^2. The gains are more obvious in the high climate version, where native woodland increases, replacing arable land that is lost to warmer temperatures and more severe droughts. This latter effect is particularly evident in England (Table 3-9), where urban areas (and particularly Greater London) benefit from urban greening (Figure 3-4).

Table 3-9 Percentage change in soil carbon stock under various scenarios of land cover change for 2060, by country

Geographic area	NEA scenarios					
	Low climate change			High climate change		
	Local Stewardship	Green and Pleasant Land	World Markets	Local Stewardship	Green and Pleasant Land	World Markets
Britain	+5.86	+8.63	–12.68	+6.30	+10.95	–11.95
England	+8.05	+13.79	–11.51	+8.72	+18.15	–10.20
Scotland	+3.01	+3.57	–13.31	+3.27	+3.87	–13.14
Wales	+8.19	+6.90	–15.35	+8.41	+8.63	–14.85

(Source: NEA 2011.)

3.1.35 In 'Green and pleasant land' (Figure 3-4 b,e), the largest gains come where the potential for habitat restoration is high: in the mountainous areas, and large parts of Wales and Scotland, where substantial reversions of enclosed farmland to semi-natural habitat are projected for this scenario. The large difference between the low and high climate scenario versions (Figure 3-4 b,e) is likely driven by an increase in semi-natural grassland under high climate, which generally replaces enclosed farmland, especially in England. Here, more than 5% more semi-natural grassland is projected for 'Green and pleasant land' under high climate change than under low climate change. This effect is also noticeable in Wales, where 'Local Stewardship' is estimated to deliver the most gains under low climate change, while 'Green and pleasant land' is estimated to deliver the most gains under high climate change (Table 3-9).

3.1.36 Overall, the estimated gains are highest under 'Green and pleasant land' (Table 3-9), with much of Britain within the range +3 to +5 ktC per km^2. The largest gains estimated under this scenario occur in South-west England, reaching +5 to +7 ktC per km^2 for the high climate version. The strength of support for agri-environment schemes and focus on sustainable management of agriculture and woodlands was perhaps borne out by the fact that no losses were estimated in the soil carbon stock under this scenario, anywhere in the UK.

3.1.37 In 'World markets' (Figure 3-4 c,f), broad-scale industrialisation of farming resulted in large net losses in soil carbon as more semi-natural and wild habitats are brought into cultivation (Table 3-9). The effects of this are most obvious in upland areas, where losses exceed 5 ktC per km^2 in the Southern Uplands, Pennines, Cumbria and the Welsh Mountains. Wales shows the highest rates of loss overall (Table 3-9), particularly in the north (Figure 3-4 c,f). This scenario does nevertheless offer some gains in urban areas, and in London (Figure 3-4 f). Unlike the other two NEA scenarios, for 'World markets' there is estimated to be less change to soil carbon stocks under high climate change than under low climate change. This is due to increased losses of arable land to higher temperatures, as adaptation capacity is low, which leads to increased drought and abandonment of unproductive land. Thus reversion to woodland or semi-natural grassland reduces estimated loss. High rates of land conversion to urban, especially in the lowlands of the South-east, appear to have less of an effect on carbon amounts, although are likely to be responsible for losses of –1 to –3 ktC per km^2 across most of lowland Britain

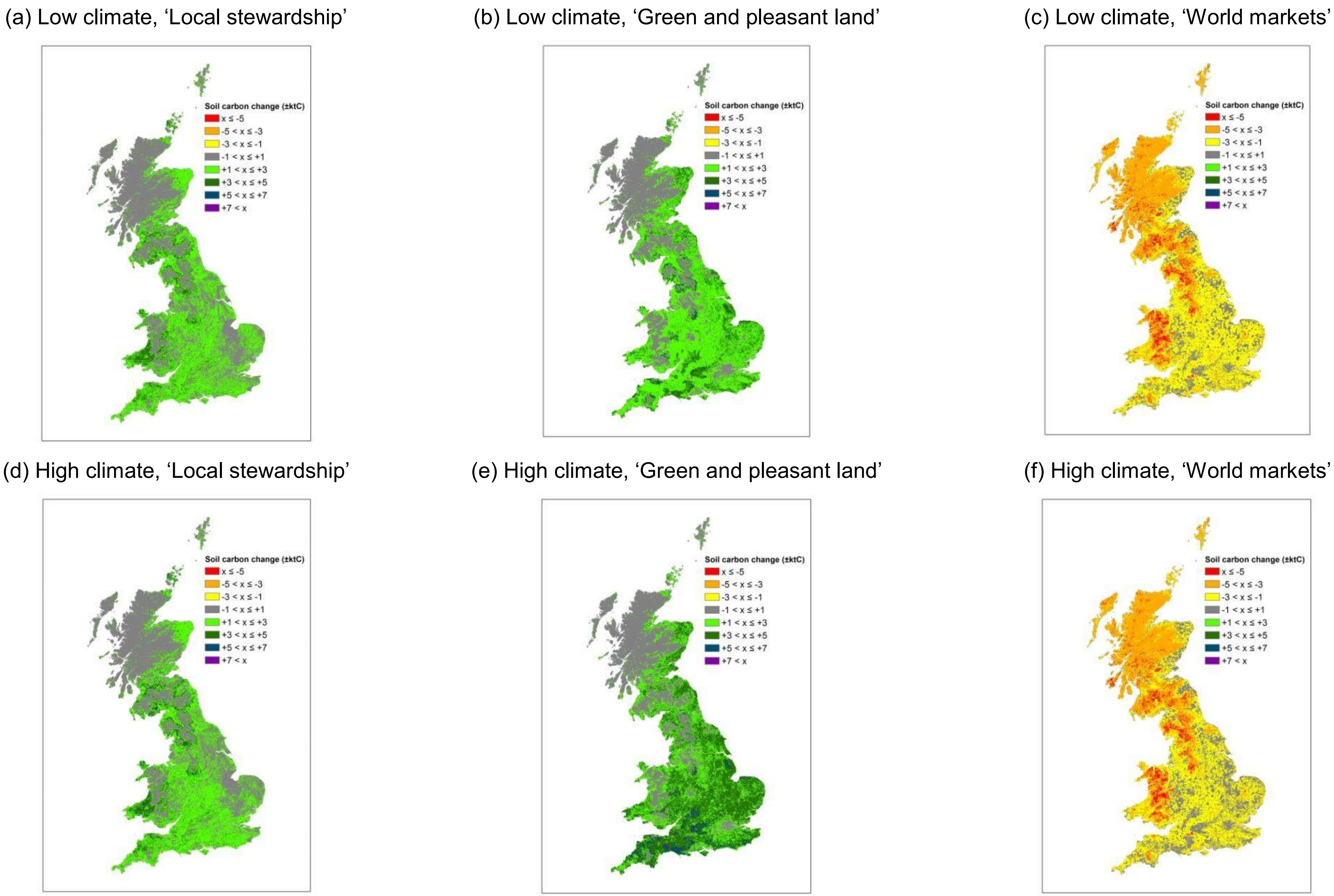


Figure 3-4 Maps of the potential change in soil carbon stock (± ktC per 1 km x 1km grid cell) arising from scenarios of future land cover change (for the year 2060)

(Source: As described in the National Ecosystem Assessment, NEA 2011. Maps are provided for low (panels a,b,c) and high (d,e,f) climate change, under three socio-economic scenarios (‘Local stewardship’, a,d; ‘Green and pleasant land’, b,e; ‘World markets’, c,f).

Sensitivity analysis based on soil depth

3.1.38 Screening out 'shallow' and 'intermediate-shallow' soil depths reduced the magnitude of estimated changes in soil carbon by a range of 1-3% across Britain (Table 3-9 and Table 3-10). Excluding shallow soils had a particularly pronounced effect on estimated changes in England and Wales. Gains were still estimated to be higher under 'Green and pleasant land' than under 'Local stewardship'. Note that losses estimated for the 'World Markets' scenario were also reduced.

Table 3-10 Percentage change in soil carbon stock under various scenarios of land cover change for 2060, by country

Geographic area	NEA scenarios					
	Low climate change			High climate change		
	Local Stewardship	Green and Pleasant Land	World Markets	Local Stewardship	Green and Pleasant Land	World Markets
Britain	+4.73	+6.99	–9.08	+5.08	+8.87	–8.49
England	+6.26	+10.90	–8.86	+6.78	+14.38	–7.82
Scotland	+2.50	+3.01	–8.42	+2.71	+3.29	–8.28
Wales	+7.07	+5.83	–12.45	+7.25	+7.28	–12.07

(Source: NEA 2011. Grid cells classified as 'shallow' or 'shallow-intermediate' on the BGS Soil Thickness map have been excluded.)

Discussion

3.1.39 The NEA scenarios provide a useful means of exploring the long-term (>50 years) soil carbon implications of land cover change. Here we used them to test how inherent uncertainties over the future socio-economic direction of the UK lead to substantially different outcomes for the UK's soil carbon stock. The multipliers calculated from Bradley *et al.* (2005), and the resultant maps we generated, should be taken as indicative estimates of the final, equilibrium change in soil carbon stock should the hypothesised land cover changes occur.

3.1.40 As we emphasise above, any changes to soil carbon stock arising from land cover changes are unlikely to happen instantly, as there will be a lag in soil carbon uptake/release following any switch between land cover types. UK NIR (2014) assume that it will take anything from 50 to 750 years for a land cover change to be reflected in the soil carbon stock of the area in question (Table 3-11), and as a rule of thumb, losses are often assumed to occur over shorter timescales than gains. This is principally due to disturbance of the soil. To reduce this disturbance, Alonso *et al.* (2012) suggested that more steady changes to habitats and soils are required (e.g. gradual felling), rather than rapid changes (e.g. clear felling). The same authors also emphasised that changes with minimal disturbance of the soil (e.g. arable to permanent or semi-natural grassland) are also likely to improve sequestration.

Table 3-11 The time taken for 99% of the change in carbon amount associated with land cover changes to take effect

Country	Time taken for change	
	Loss	Gain
England	50 – 150 years	100 – 300 years
Wales		
Scotland		300 – 750 years

(Source: adapted from UKNIR 2014, Table A 3.6.17)

3.1.41 Some of the areas of high loss or gain reported on our maps may also be unlikely (or indeed impossible) due to existing site constraints. Our sensitivity analysis showed that the magnitude of both gains (range +6 to +11% across Britain) and losses (–12 to –13%) we estimated were considerably reduced by filtering out shallow soils from the analysis. There will also be other site constraints that were not considered in the scenario modelling methodology; for example, the 'World Markets' policy of widespread urbanisation in the uplands- responsible for a substantial part of the losses associated with this scenario- sees urban fractions increase substantially in areas currently designated for conservation and protected from development.

3.1.42 The difference in the risk from land cover change between the high and low climate versions was small, and this is reflective of the land cover data themselves: NEA reported that the variation due to assumed socio-economic scenario was much larger than that due to the climate versions of each scenario. In particular, Local Stewardship assumes a high level of adaptation to climate change, and in this case adaptations reduce the effect of (and thus differences between) the scenario changes. It should be emphasised that we only test for an indirect effect of climate change, and not a direct, physical effect of warming or precipitation changes on carbon levels. Thus the combined (direct and indirect) effects of climate change on soil carbon stocks are likely to be larger than Figure 3-4 and Table 3-9 suggest.

3.1.43 Climate change can affect soil carbon levels directly via two principal means- productivity and decomposition. Were climate change to accelerate rates of decomposition, release of carbon from the soil to the atmosphere would occur. However, increased productivity would result in enhanced carbon uptake by plants, and could act to offset or eclipse losses from decomposition. A number of laboratory (e.g. Karhu et al. 2014), modelling (e.g. Gottshalk et al. 2012), and field experiments (e.g. Giardina et al. 2014) have determined warming-related effects on both decomposition and productivity, but crucially, none have resolved the question of what the net effect of climate change will be (see Davidson and Janssens 2006 for a review). This question may not have a simple answer (Gottshalk et al. 2012). It is also likely that a number of other factors (e.g. soil moisture, erosion) both a) exert controls on soil carbon budgets, and b) also affect productivity and decomposition. These factors will also be affected by climate change.

3.1.44 Given this uncertainty, research into the degree to which land use/cover changes result in carbon sequestration is ongoing. Although the means of carbon sequestration for some land cover changes are more obvious (e.g. peatland restoration for changes to 'Mountains, Moors and Heath'), in other cases the mechanisms for sequestration for particular land cover changes are unclear. Thus, they are likely to be overestimates of the gains and losses in carbon associated with these changes. There are also cases when empirical data on certain changes do not (yet) reflect our potential, equilibrium changes. For example, we assume a change of 'Arable' to 'Woodland' will result in a net increase in soil carbon of 13 kg / m^2, or an increase of ~ 108% (Table 3-6), but a recent meta-analysis estimated the likely change associated with 'cropland' to 'forest' to be within the range 15 – 40% (Laganière et al. 2010). The implications of planting are also likely to depend on the balance of evergreen or broadleaf trees being planted (Forest Research 2012). Long-term monitoring of all types of land use change in a range of contexts should improve our knowledge of the magnitude of its effect on carbon values.

3.1.45 There are also cases where the current evidence on carbon fluxes arising from the change is conflicting: for example, although changes in land use/land cover are commonly perceived as beneficial to carbon stocks when afforestation has taken place, Guo and Gifford's (2002) meta-analysis illustrated that some changes from open habitats to forest can result in losses to carbon stock. Whereas Guo and Gifford (2002) suggested that a switch from pasture to plantation reduces the soil carbon content by 10%, the data in Laganière *et al.* (2010) suggested that a small soil carbon increase results from conversion from 'Pasture' to 'Forest'. The same Laganière *et al.* (2010) study did also highlight that the biggest benefits to soil carbon stocks from afforestation are available in temperate maritime climates, such as that of the UK, but as Guo and Clifford (2002) specified, this afforestation must be appropriately targeted (with soil carbon in mind).

Changes in vegetation carbon stocks arising from land cover change

Justification

3.1.46 The role of LULUCF as a possible carbon sink for inventory reporting under the UNFCCC provides an incentive for signatories to 'lock up' carbon in afforestation schemes. However, almost any change in land cover will have an effect on the UK's stocks of vegetation carbon. If managed appropriately, changes in these stocks could make an important contribution towards the UK reducing net emissions.

3.1.47 In the NEA dataset, some of the land cover changes involve intentional planting of woodland, e.g. for agri-environment schemes in 'Green and pleasant land', or to meet local resource demand in 'Local stewardship'. Other land cover changes to woodland are more an indirect consequence of planning decisions, e.g. the abandonment of less productive arable land (presently subject to subsidy) under 'World markets'. It is important to capture these latter effects also, as their implications for vegetation carbon stock can be substantial.

Data

3.1.48 As per the previous section, we adopted the future land cover scenarios generated for the National Ecosystem Assessment (NEA 2011). These represent future distributions of land cover (2060) under three socio-economic scenarios: 'Local stewardship', 'World Markets' and 'Green and pleasant land'. These socio-economic scenarios come in two versions: 'high' and 'low' climate change, in which the indirect effect of climate change is captured (e.g. via increased prevalence of droughts under 'high' climate change), and different levels of adaptation to climate change are assumed.

Methods

3.1.49 We used the density values set out for 'vegetation groups' in Milne and Brown (1997, given in Table 15 of Milne and Brown 1997, repeated in Table 3-12). These values provided the basis for estimating the amount of carbon in each 1km grid cell under each NEA scenario (n=3, 'Local stewardship', 'Green and pleasant land', 'World Markets') and climate (n=2, 'high' and 'low') combination. Here, as before, the 'vegetation groups' of Milne and Brown (1997) do not match the broad habitats of the NEA (2011) exactly; we thus used the scheme in Table 3.13 to 'translate' the density values.

Table 3-12 Vegetation carbon multipliers for land cover changes described in the NEA scenario data

Milne and Brown (1997) vegetation group	Vegetation carbon density (t / hectare)
Semi-natural	1.66
Woodland	36.84
Agricultural	0.97

(Source: NEA 2011. Multipliers were derived from the carbon densities of vegetation groups set out in Milne and Brown (1997)).

3.1.50 As before, it must be emphasised that the resultant estimates represent the equilibrium carbon change following the change in land cover, and that such changes could take longer than 50-60 years to take effect. Typically, carbon uptake in trees is highest after peak timber increment (i.e. in the period after canopy closure), while in forest stands managed for carbon, the stand can take ~ 100 years to reach maturity and approach its maximum carbon capacity (Forest Research 2012). Thus, as for the soil carbon changes, the estimates we calculate must be interpreted as final, equilibrium changes in vegetation carbon resulting from the land cover changes that the NEA dataset documents.

3.1.51 NEA (2011) also undertook analyses of the implied change in vegetation carbon arising from changes in the cover of the NEA broad habitats, using the same density estimates set out in Milne and Brown (1997) for semi-natural and agricultural vegetation groups (see Table 3-13 for the alignment of Milne and Brown (1997) vegetation groups and NEA (2011) broad habitats). However, densities of 58.59 t/ha and 21.27 t/ha were used for broadleaf and coniferous forest types, respectively, whereas we use the 36.84 t/ha value reported alongside the semi-natural and agricultural values in Table 15 of Milne and Brown 1997. Although initial work suggested that broadleaf had a higher mean carbon density than coniferous (Milne and Brown 1997), a more recent summary of the latest research published after NEA (2011) has concluded that the evidence is unclear (Forest Research 2012). We thus took the decision to adopt the average for woodland as a whole.

Table 3-13 Scheme to match the NEA broad habitats to the Milne and Brown vegetation groups.

Milne and Brown (1997) vegetation group	NEA broad habitat (NEA 2011)
Semi-natural	Mountains, Moors, Heaths
	Semi-natural grasslands
Woodland	Woodlands
Agricultural	Arable
	Improved grassland

(Source: NEA 2011 and Milne and Brown 1997).

Results

3.1.52 The three scenarios again generate substantial differences in estimated carbon amounts, and in the case of 'Local stewardship' and 'World markets', these estimates are notably different under 'high' and 'low' climate change (Table 3-14, Figure 3-5). The difference between 'high' and 'low' is most evident in the South and South-east of England, where the impacts of climate change are projected to be large (NEA 2011). Unlike the soil carbon analyses, all three scenarios are associated with both losses and gains, at both levels of climate change; losses are typically focussed in the Galloway, Borders and Kielder areas of existing forest stock (Figure 3-5, a-f), while gains are estimated for lowlands in the south and east of Britain (Figure 3-5, b,d,e,f; also England in Table 3-14).

Table 3-14 Percentage change in vegetation carbon stock under various scenarios of land cover change for 2060 (NEA 2011), by country

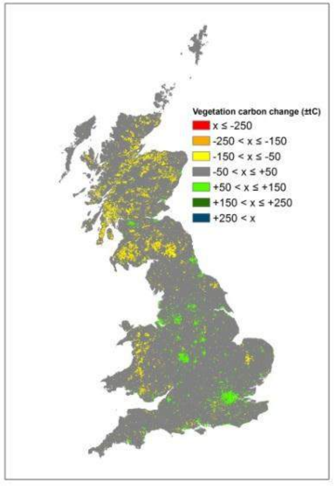
Geographic area	NEA scenarios					
	Low climate change			High climate change		
	Local Stewardship	Green and Pleasant Land	World Markets	Local Stewardship	Green and Pleasant Land	World Markets
Britain	–0.25	+28.33	–7.96	+6.15	+23.23	–3.16
England	+1.49	+48.91	–9.18	+13.30	+40.06	–0.29
Scotland	–2.39	+4.84	–5.17	–1.49	+4.39	–4.59
Wales	–0.08	+5.63	–3.37	+0.48	+4.26	–2.93

3.1.53 The land cover changes associated with ‘local stewardship’ have a minimal effect on vegetation carbon stocks under low climate change (Figure 3.6a), with modest gains (+ 50 to +150 tC per km^2) in urban areas (urban greening), and similar levels of loss (– 50 to –150 tC per km^2) in areas of existing forest stock, particularly in Scotland. This balance of gain and loss results in almost no net change in vegetation carbon stocks across Britain (–0.25% loss, Table 3.14). However, under high climate change, the loss of arable land to droughts and warmer temperatures sees large parts of lowland Britain converted to native woodland (Figure 3-5 d), resulting in a net increase in stock across Britain (+6.15% gain, Table 3.14). These gains are most pronounced in Dorset and along the south coast.

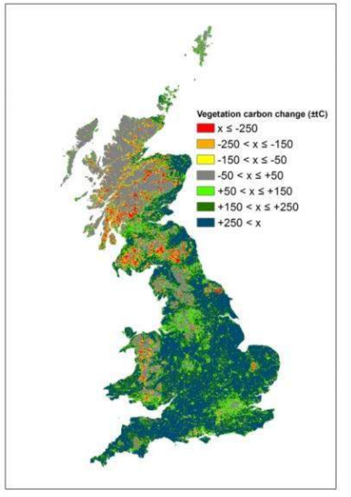
3.1.54 The broad-scale afforestation programme associated with ‘Green and pleasant land’ resulted in predictably large increases in estimated vegetation carbon across much of lowland Britain, particularly in England (Figure 3-5 b,e; Table 3-14). Many of these gains exceed 250 tC per km^2. Areas of more modest gains include National Parks and Areas of Outstanding Natural Beauty, where habitat restoration schemes are either established or expanded to target semi-natural habitats instead. As for ‘Local stewardship’, most of the Highlands of Scotland is estimated to experience little or no change in vegetation carbon stocks, and estimated gains in Scotland are smaller overall (Table 3-14). Unlike ‘Local stewardship’ however, less focus on local provision of resources (e.g. timber) sees low estimated gains in urban areas, with areas of high population density (London, Birmingham, Manchester) estimated to have little/no net change (Figure 3-5 b,e).

3.1.55 Because much of the vegetation carbon is contained within forests, drops in the cover of ‘Semi-natural grassland’ and ‘Mountain, Moors and heaths’ associated with ‘World Markets’ have a less deleterious effect on vegetation carbon stocks than on soil carbon stocks, where these habitats are critical. Under both ‘low’ and ‘high’ climate change, there is a notable effect of abandonment in upland areas, where the lack of management results in increased succession of woodland: this effect is strongest in the Central Highlands of Scotland where gains of +150 to +250 tC per km^2 are estimated. Similar to ‘Local stewardship’, loss of arable land to drought and high temperatures sees estimated vegetation carbon gains across much of the South of England, and these gains are large enough to change the estimated net change in vegetation carbon stock for England from –9.18% under low climate change to –0.29% under high climate change. The changes in the net estimates for Wales and Scotland under high climate change (vs. low climate change) are smaller (approximately 0.5%).

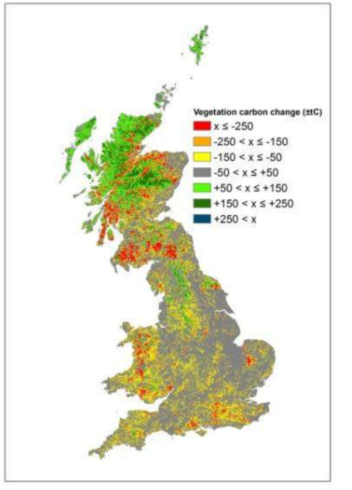
(a) Low climate, ‘Local stewardship’.



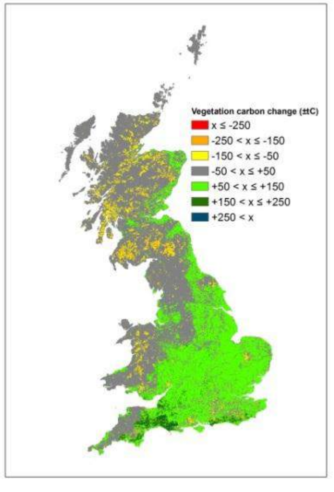
(b) Low climate, ‘Green and pleasant land’



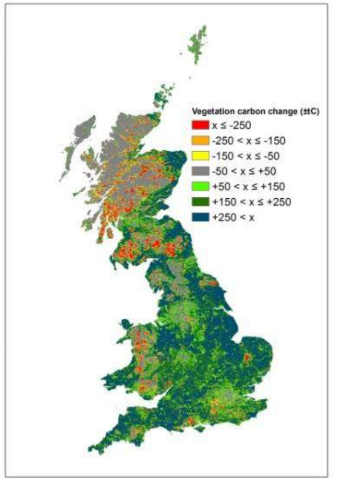
(c) Low climate, ‘World markets’.



(d) High climate, ‘Local stewardship’



(e) High climate, ‘Green and pleasant land’



(f) High climate, ‘World markets’

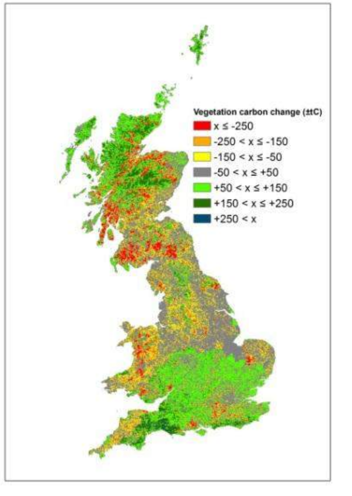


Figure 3-5 Maps of the potential change in vegetation carbon stock (± tC per 1 km x 1km grid cell) arising from scenarios of future land cover change (for the year 2060).

Source: NEA (2011). Maps are provided for low (panels a,b,c) and high (d,e,f) climate change, under three socio-economic scenarios (‘Local stewardship’, a,d; ‘Green and pleasant land’, b,e; ‘World Markets’, c,f).

Discussion

3.1.56 We used the NEA scenario dataset to provide an indicative idea of the potential changes in vegetation carbon arising from three scenarios of socio-economic change in the UK. The implications of these scenarios were quite diverse, ranging from a –3% to –8% net loss across Britain under 'World Markets', to a +23% to +28% net gain under 'Green and pleasant land'. 'Local stewardship' had a relatively small effect, ranging from approximately no net loss/gain under low climate change, to a +6% gain under high climate change. Scenario-related uncertainty was thus higher for vegetation carbon stocks than for soil carbon stocks, with socio-economic direction exerting a larger influence on their status as a source or sink of carbon. Although this means that a greater potential exists for changing this status via changes to land management, it must be emphasised that much more carbon is stored in the soil than in the vegetation, even inside forests (Forest Research 2012).

3.1.57 It is useful to compare our analyses to NEA (2011, Figure 25.21 in Chapter 25), who also used Milne and Brown (1997) values for vegetation carbon content to estimate changes in vegetation carbon density due to the NEA scenarios (under low climate change only). Similar to our analyses, they found assumed carbon estimates to be highly variable across the scenario set, and produced similar outputs for the scenarios we adopt. Similar to our estimates, NEA (2011) estimates showed little or no change in vegetation carbon density under 'Local stewardship', losses in existing forested areas and uplands under 'Green and Pleasant Land', and the gains we estimated for the Highlands of Scotland and northern England under 'World Markets'. Overall, there is good agreement between the maps.

3.1.58 It must again be emphasised that our estimates represent the carbon gains possible once changes to land cover have taken effect. It is obvious that loss of woodland carbon can proceed quickly, however, the extent to which carbon accumulates in new forest depends upon management, climate, and the species concerned (Forest Research 2012). Above, we cited the estimate of 100 years for woodland managed specifically for carbon to reach maturity, and approach maximal carbon storage; although management can also be adjusted to encourage rapid carbon uptake in the establishment phase (Forest Research 2012). Either of these approaches will undoubtedly take longer than conversion to other land cover types via deforestation, and therefore, as per our soil carbon analyses, losses will proceed faster than gains.

3.1.59 The gains estimated under 'Green and pleasant land' were large, particularly in England (+40% to +49%). These numbers appear large, but the afforestation rates that drive them are not unrealistic (e.g. increase of 5% to 7% in area occupied by woodland in England). However, planting must be managed strategically, while disturbance and drainage must be carefully handled to minimise soil carbon losses (Forestry Commission 2012). It must also be conducted within the wider guidelines of the UK Forestry Standard, under which sustainable management practices for biodiversity, climate change, historic environment, landscape, people, soil and water are encouraged, and in some cases, required by law (Forestry Commission 2011). Managing for multiple benefits will ensure that the benefit we derive from the various ecosystem services derived from forests (including provision of an equable climate) is maximised.

3.1.60 The influence of other threats to woodland carbon stock must also be considered. For example, if the spread of ash dieback (*Hymenoscyphus pseudoalbidus*) proceeds as expected (90% infection, 60% mortality) then this could lead to the loss of 5Mt of carbon, or 2.3% of the vegetation carbon store based on the FC estimate above (Reay 2013). This may not result in a net 5Mt flux directly to the air or water environments however, as increased input to the soil and litter layers will undoubtedly occur, and there remains a possibility that the ash is used as a replacement fuel. Similar to the St Jude storm, potential harms are felt more keenly when their effects preclude any further use of the timber, and it may be that the biggest threat to vegetation carbon stocks from climate change comes from storms like St Jude, which are projected to become more likely under climate change (Forestry Commission 2014b).

Valuation

3.1.61 The aim of this section is to develop estimates of the monetary value of changes in carbon stored in soil and vegetation stocks under a range of possible climate scenarios. In order to align with the UK NEA scenarios, the change in carbon is measured over the period 2010 to 2060. The UK NEA scenarios were developed for Great Britain and so do not include Northern Ireland. As such, estimates of the change in carbon stores in Northern Ireland are excluded from the assessment due to a lack of available data.

Quantifying changes in soil carbon

3.1.62 The estimates of changes in soil carbon included two key factors: (1) the change in soil carbon due to soil erosion; and (2) the change in soil carbon due to land use change. The total change in soil carbon was based on an aggregation of these two factors.

3.1.63 Estimates of baseline soil carbon in 2010 were based on the figures provided in Bradley et al. (2005)[5] for soil up to a depth of 100 cm, while estimates of the annual change in soil carbon from the top 100 cm of soil due to erosion were developed using the Pesera model (see Table 3-15).

Table 3-15 Baseline soil carbon in 2010 and annual change in carbon due to erosion in soils up to a depth of 100 cm (MtC and $MtCO_2e$)

Region	Baseline soil carbon (MtC)	Annual change (MtC)*	Annual change ($MtCO_2e$)*
England	1,740	-0.0446	*-0.1637*
Scotland	2,187	-0.0347	*-0.1274*
Wales	340	-0.0019	*-0.0070*
Great Britain	4,267	-0.0813	*-0.2981*

**Results presented to 4 decimal places*

3.1.64 Changes in soil carbon due to land use change were estimated based on the scenarios developed in the UK NEA,[6] in particular for the local stewardship, green and pleasant land, and world markets scenarios. For each of these three scenarios a 'low' emissions and 'high' emissions scenario is presented giving a total of six individual scenarios. Using these six scenarios, the percentage changes in soil carbon stocks over the period 2010 to 2060 resulting from land use change are set out in Table 3-16.

Table 3-16 Change in soil carbon stocks from 2010 to 2060 as a result of land use change (%)

Region	Local stewardship (low)*	Local stewardship (high)*	Green and pleasant land (low)*	Green and pleasant land (high)*	World markets (low)*	World markets (high)*
England	8.05%	8.72%	13.79%	18.15%	-11.51%	-10.20%
Scotland	3.01%	3.27%	3.57%	3.87%	-13.31%	-13.14%
Wales	8.19%	8.41%	6.90%	8.63%	-15.35%	-14.85%
Great Britain	5.86%	6.30%	8.63%	10.95%	-12.68%	-11.95%

**Results presented to 2 decimal places*

3.1.65 The annual percentage change in carbon stocks due to land use change was estimated by dividing the total values by the total period of time (i.e. 50 years). This was then combined with the baseline values of soil carbon to estimate the annual change in soil carbon stocks in millions

[5] Bradley et al. (2005), 'A soil carbon and land use database for the United Kingdom', Soil Use and Management Volume 21, Issue 4, pages 363–369, December 2005

[6] Haines-Young (2011), 'Chapter 25: The UK NEA Scenarios: Development of Storylines and Analysis of Outcomes', UK NEA.

of tonnes of carbon. The results are set out in Table 3-17, although it is important to note that, in practice, it may take longer than 50 years for land use change to have an effect on soil carbon density and so the change may be overstated.

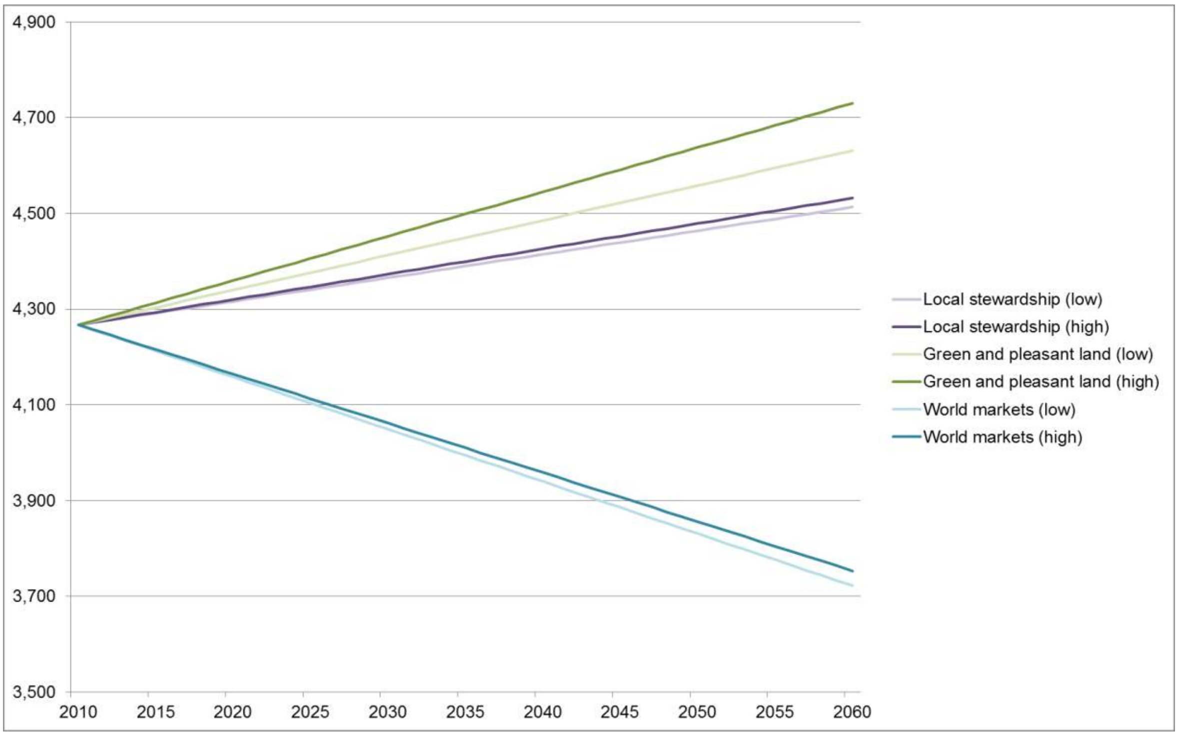
Table 3-17 Annual change in soil carbon stocks from 2010 to 2060 as a result of land use change (MtC and $MtCO_2e$)

Region	Local stewardship (low)	Local stewardship (high)	Green and pleasant land (low)	Green and pleasant land (high)	World markets (low)	World markets (high)
England						
MtC	2.8009	3.0346	4.7974	6.3172	-4.0050	-3.5488
$MtCO_2e$	*10.2700*	*11.1269*	*17.5905*	*23.1631*	*-14.6850*	*-13.0123*
Scotland						
MtC	1.3184	1.4295	1.5635	1.6913	-5.8239	-5.7459
$MtCO_2e$	*4.8341*	*5.2415*	*5.7328*	*6.2014*	*-21.3543*	*-21.0683*
Wales						
MtC	0.5567	0.5717	0.4689	0.5870	-1.0435	-1.0101
$MtCO_2e$	*2.0412*	*2.0962*	*1.7193*	*2.1523*	*-3.8262*	*-3.7037*
Great Britain						
MtC	4.9996	5.3785	7.3653	9.3469	-10.8219	-10.1958
$MtCO_2e$	*18.3319*	*19.7212*	*27.0061*	*34.2720*	*-39.6803*	*-37.3846*

**Results presented to 4 decimal places*

3.1.66 The total annual change in soil carbon stocks was then estimated by aggregating the annual change due to soil erosion and to land use change. Figure 3-6 provides an overview of the total change in soil carbon across Great Britain for each of the six scenarios.

Figure 3-6 Change in soil carbon stocks across Great Britain from 2010 to 2060 as a result of soil erosion and land use change combined for the six UK NEA scenarios (MtC)



Quantifying changes in vegetation carbon

3.1.67 Estimates of the change in vegetation carbon stocks were based entirely on the six land use change scenarios described above. The same approach was used to quantify the annual change in carbon stocks and the results are set out in Table 3-18.

Table 3-18 Annual change in vegetation carbon stocks from 2010 to 2060 as a result of land use change (MtC and $MtCO_2e$)

Region	Local stewardship (low)	Local stewardship (high)	Green and pleasant land (low)	Green and pleasant land (high)	World markets (low)	World markets (high)
England						
MtC	0.0315	0.2802	1.0308	0.8443	-0.1936	-0.0060
$MtCO_2e$	*0.1155*	*1.0274*	*3.7796*	*3.0958*	*-0.7099*	*-0.0220*
Scotland						
MtC	-0.0409	-0.0255	0.0826	0.0750	-0.0883	-0.0784
$MtCO_2e$	*-0.1500*	*-0.0935*	*0.3029*	*0.2750*	*-0.3238*	*-0.2875*
Wales						
MtC	0.0000	0.0020	0.0250	0.0190	-0.0150	-0.0130
$MtCO_2e$	*0.0000*	*0.0073*	*0.0917*	*0.0697*	*-0.0550*	*-0.0477*
Great Britain						
MtC	-0.0109	0.2620	1.2067	0.9897	-0.3391	-0.1346
$MtCO_2e$	*-0.0400*	*0.9607*	*4.4246*	*3.6289*	*-1.2434*	*-0.4935*

**Results presented to 4 decimal places*

3.1.68 Estimates of the baseline quantity of vegetation carbon stocks were based on the figures presented in Forestry Commission (2014)[7] as set out in Table 3-19.

Table 3-19 Baseline vegetation carbon in 2010 (MtC and $MtCO_2e$)

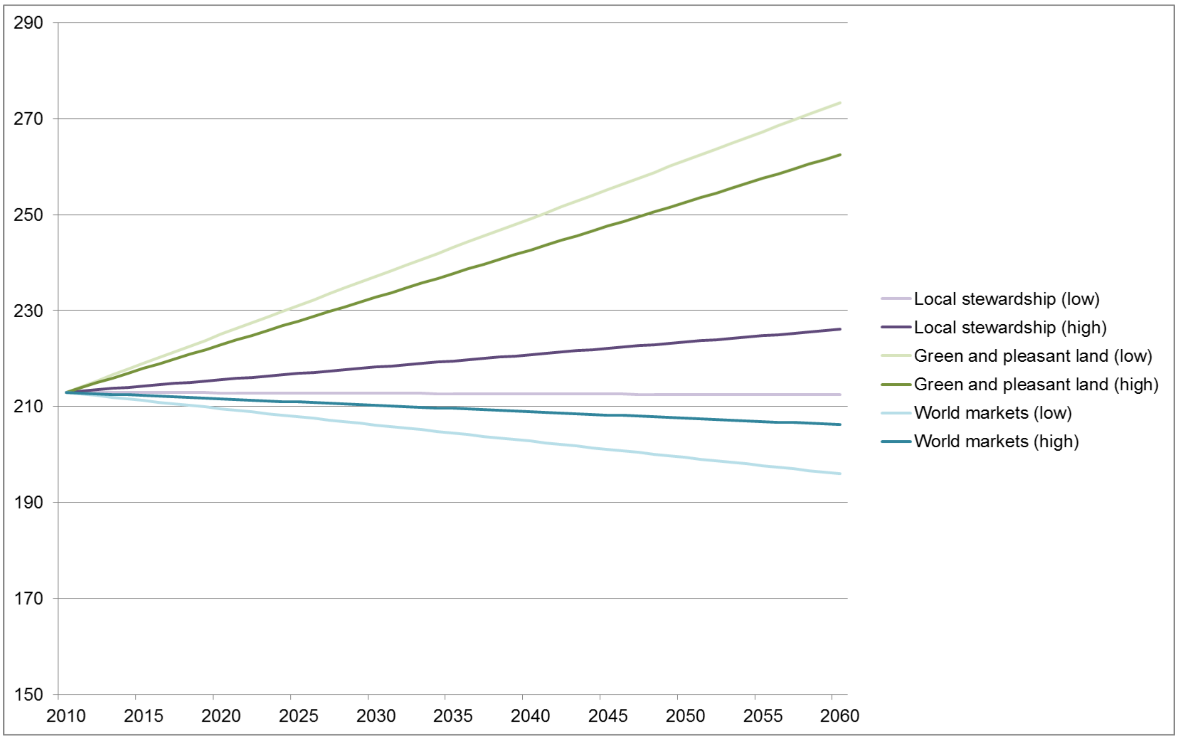
Region	Baseline vegetation carbon (MtC)	Baseline vegetation carbon ($MtCO_2e$)
England	105.3880	*386.4227*
Scotland	85.4410	*313.2837*
Wales	22.1430	*81.1910*
Great Britain	212.9720	*780.8973*

**Results presented to 4 decimal places*

3.1.69 Figure 3-7 provides an overview of the total change in vegetation carbon across Great Britain for each of the six scenarios.

[7] Forestry Commission (2014), 'Carbon in live woodland trees in Britain National Forest Inventory Report'.

Figure 3-7 Change in vegetation carbon stocks across Great Britain from 2010 to 2060 as a result of land use change for the six UK NEA scenarios (MtC)



Valuation basis

3.1.70 There is no nationally or globally agreed value for carbon (or CO_2e) and estimates range from -$6.6 to $2,400 (-£4.3 to £1,580) per tonne (Tol 2013). Estimating the value of non-market GHG emissions is challenging for two main reasons (Abson et al. 2010). First, there is, as yet, no definitive relationship between emissions and climate change. Moreover, there is considerable uncertainty regarding the relationship between climate change and its impacts on the economy, as the nature and significance of impacts depend on socio-technological responses to changes in the climate.

3.1.71 Second, when forecasting carbon values, the societal cost associated with the emission of an additional tonne of carbon is dependent on how many tonnes of carbon have previously been emitted (and abated), the eventual concentrations at which carbon dioxide is stabilised in the atmosphere, and the emissions trajectory adopted to achieve this stabilisation (Bateman et al. 2011). As such, future carbon prices depend upon the emission and climate scenarios upon which they are based.

3.1.72 There are two main approaches to carbon pricing: the social cost of carbon (SCC); and the marginal abatement cost of carbon (MACC). These are explained in detail in Price et al., (2007) and Bateman et al (2014) and summarised in the box below.

Box 3.1 Approaches to valuing carbon

The **social cost of carbon (SCC)** may be defined as the cost of total global damages caused by an incremental unit of carbon emitted today, summed over its entire time in the atmosphere, and discounted to present value terms (Price et al., 2007). However, there are several challenges with using SCC including:

- Estimates of total damage vary widely due to the extent of uncertainty surrounding ‘fat tails’ (Pycroft et al., 2011), environmental tipping points (Lenton et al., 2008; Weitzman, 2009), and the biosphere’s precise response to atmospheric carbon (IPCC, 2007).
- Estimates of SCC are particularly sensitive to the discount rate used, as well as a multitude of other assumptions regarding consumption growth rates, projected CO_2 emissions, the carbon cycle, and environmental sensitivity to CO_2 concentrations and temperature change.

- They have attracted intense criticism (see Pindyck, 2013 and Stern, 2013) because of the fundamental problems with the underlying models used to generate the estimates.

The **marginal abatement cost (MAC)** approach estimates the cost to polluters of reducing emissions by an incremental amount. This entails setting an emissions cap or reductions target relative to some base level, and then estimating the cost of meeting it (Dietz and Fankhauser, 2010). While there are still uncertainties with these estimates, the range of uncertainty is much narrower than that around the SCC (Bateman et al., 2014). However, MAC curves remain sensitive to assumptions regarding the lifetime of technologies, their investment and operating costs, and, of the choice of discount rate (Kesicki and Ekins, 2011). Moreover, unlike the SCC, this approach obtains values implied by a specific target, but it cannot tell us anything about whether the chosen target (and its implicit price) is the 'right' one. The MAC approach could therefore yield carbon values that are well below or above above the 'true SCC', causing us to under- or over-abate relative to the (unknown) true social optimum.

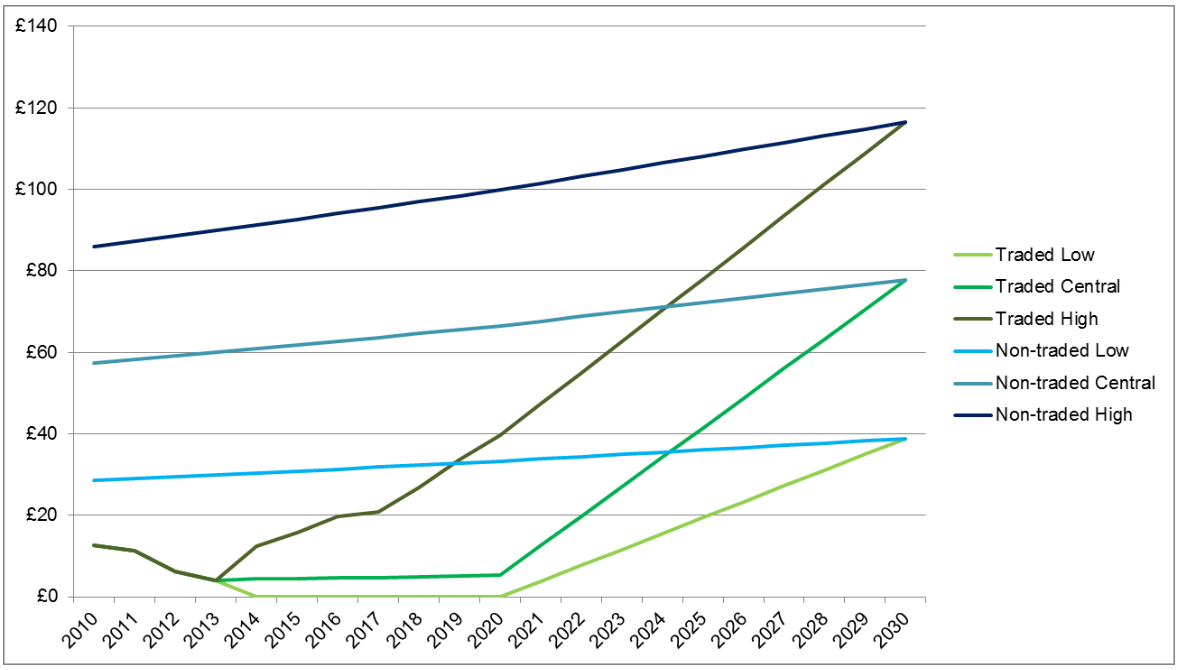
3.1.73 Given the relative strengths and weaknesses of each of the approaches, the most appropriate and robust method of valuing changes in carbon stocks is likely to be the values published by UK Department of Environment and Climate Change (DECC)[8] for use in UK policy appraisal (see Bateman et al., 2014). The DECC (2014) values are derived on the basis of the costs of implementing the level of abatement necessary to achieve the targets set out in the UK Climate Change Act and the first report of the Committee on Climate Change (2008) to the UK Government. As such, this approach avoids the complications and uncertainties associated with quantifying the impacts of GHG emissions.

3.1.74 These marginal abatement costs are derived from the UK's EU and UN commitments which require a reduction in total UK emissions of 80% of their 1990 level by 2050, with interim targets of 26% and 12.5% reductions relative to 1990 levels by the years 2020 and 2012, respectively. The DECC estimates provide low, central and high cost estimates for GHG emissions up to the year 2100.

3.1.75 The EU Climate and Energy Package (December 2008) introduced separate emissions reduction targets for the traded sector (that is those emissions covered by the EU Emissions Trading System, (ETS), and for the non-traded sector (those emissions not covered by the EU ETS). The presence of separate targets in the Traded and Non-Traded sectors implies that emissions in the two sectors are essentially different commodities. As such, changes in emissions which occur in the traded sector are valued at the Traded Price of Carbon (TPC), whereas changes in emissions in the non-traded sector are valued at the Non-Traded Price of Carbon (NTPC). A comparison of the traded and non-traded carbon prices during the period 2010 to 2030 is set out in Figure 3-8.

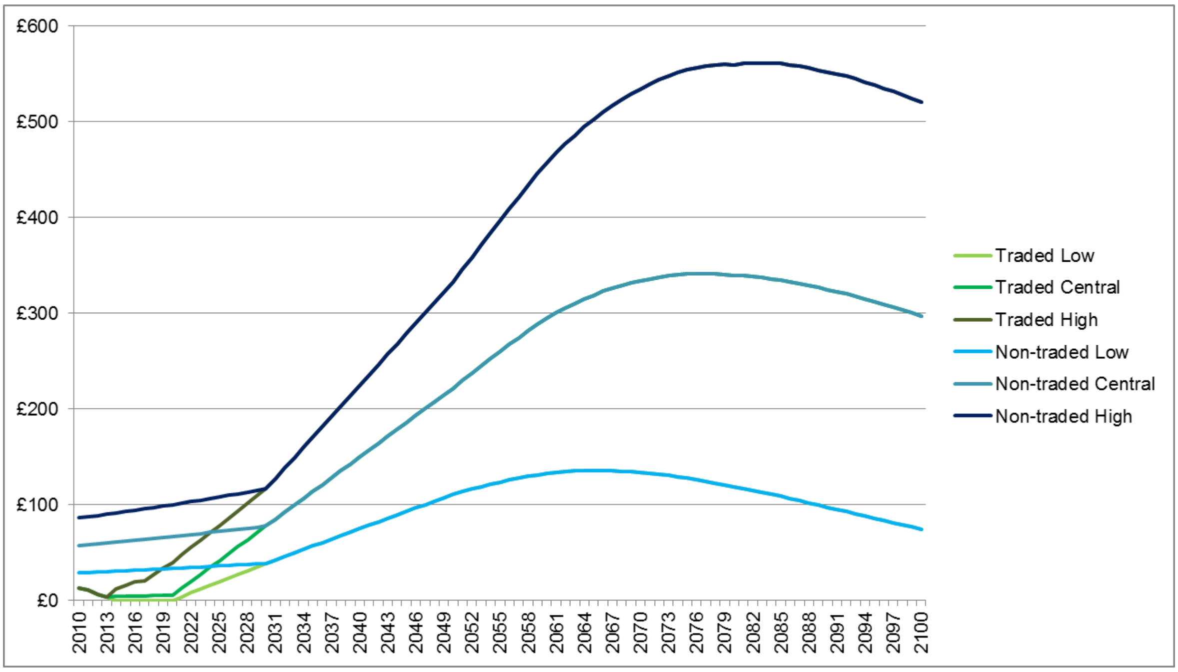
[8] DECC (2014), 'Green Book supplementary guidance: valuation of energy use and greenhouse gas emissions for appraisal', [online] https://www.gov.uk/government/publications/valuation-of-energy-use-and-greenhouse-gas-emissions-for-appraisal

Figure 3-8 Comparison of DECC traded and non-traded carbon prices during the period 2010 to 2030 (£/tCO_2e)



3.1.76 According to DECC modelling estimates, the traded and non-traded prices are different in the short-term, but are projected to converge, becoming equal in 2030 and remaining so in further years. This is based on the assumption that there will be a functioning global carbon market by 2030. An overview of these long term trends up to 2100 is set out in Figure 3-9.

Figure 3-9 Comparison of DECC traded and non-traded carbon prices during the period 2010 to 2010 (£/tCO_2e)

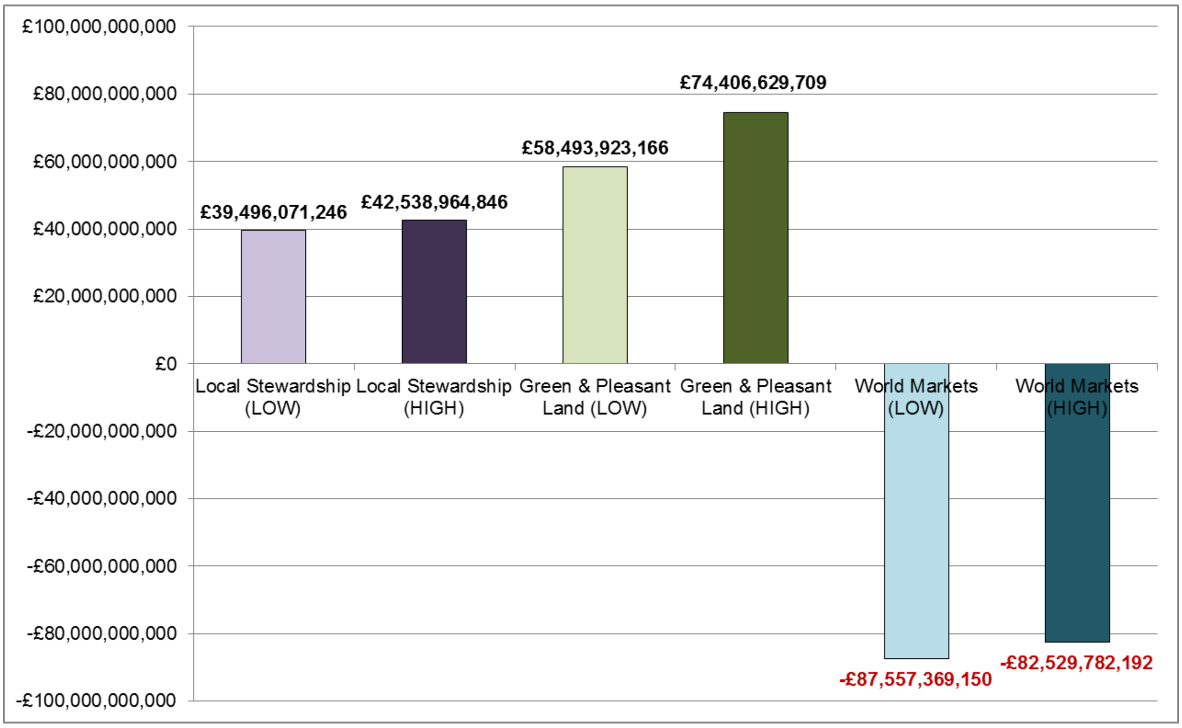


3.1.77 In order to estimate the value of the change in soil and vegetation carbon stocks over the period 2010 to 2060, the estimates of the change in carbon were converted from millions of tonnes to carbon to tonnes of CO_2 equivalent (tCO_2e).[9] The annual change in tonnes of CO2 equivalent was then multiplied by the central non-traded DECC carbon prices for the period 2010 to 2060. Following the guidance set out in The Green Book,[10] these values were then discounted using a rate of 3.5% for the first 30 years and 3.5% thereafter in order to estimate the Net Present Value (NPV) of the change.

3.1.78 The results of this approach suggest that the total value of the change in soil carbon stocks across Great Britain over the period 2010 to 2060 ranges from a low of -£87.6 billion in the world markets (low emissions) scenario, to a high of £74.4 billion in the green and pleasant land (high emissions scenario). A comparison of the value estimates is presented in Figure 3-10, all values are presented in 2014 prices.

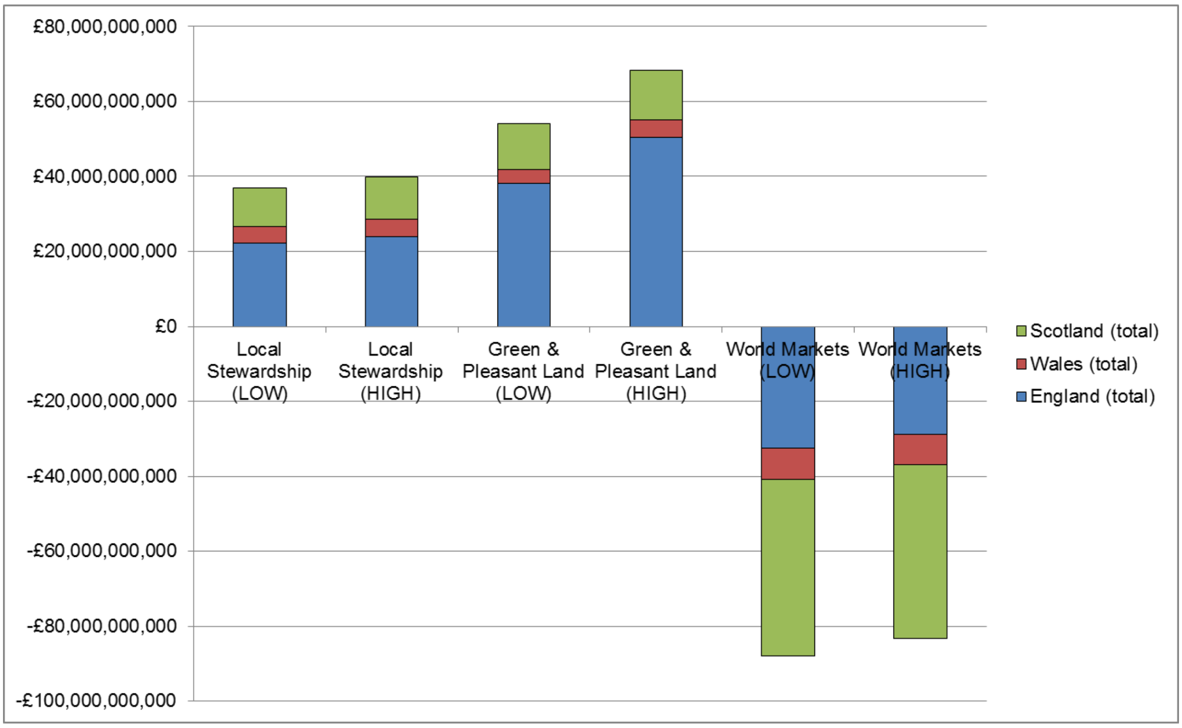
[9] The conversion was based on multiplying the estimates by a factor of 1,000,000 then multiplying the resulting estimate by the molecular weight of carbon relative to oxygen in a molecule of CO_2 i.e. 44:12.
[10] HM Treasury (2014), 'The Green Book: appraisal and evaluation in central government'

Figure 3-10 Comparison of the NPV of changes in soil carbon stocks across Great Britain from 2010 to 2060



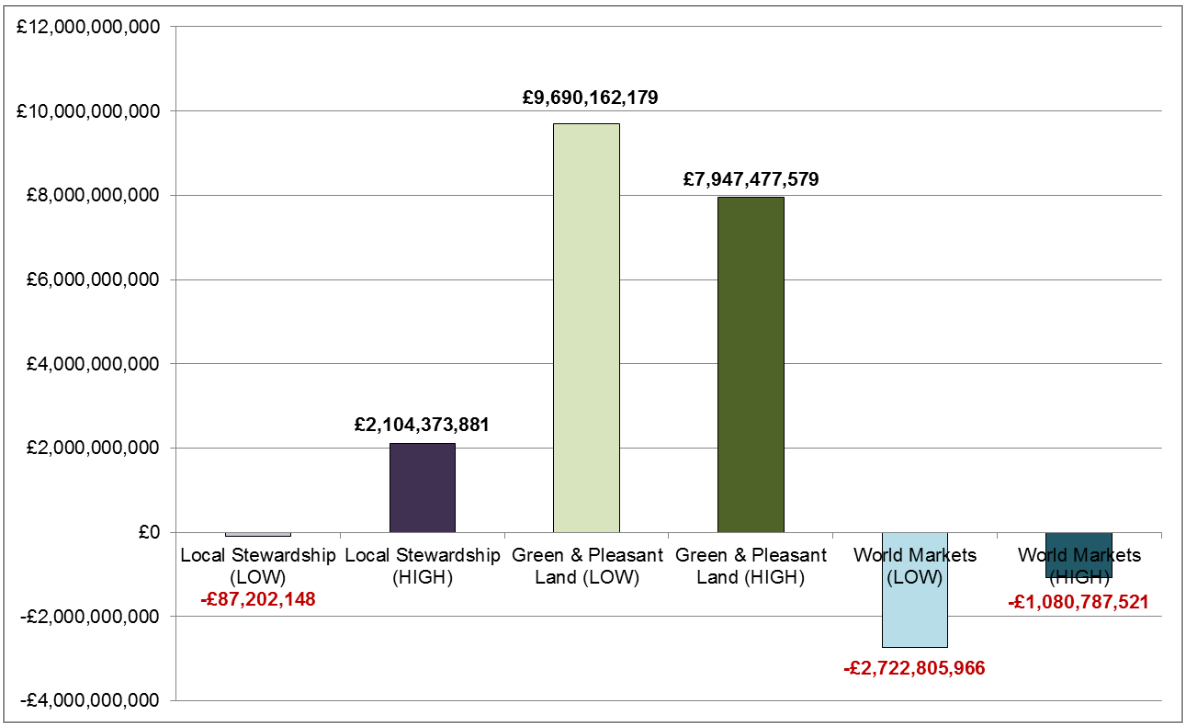
3.1.79 Breaking down the estimates at a national level reveals an greater range of variation as set out in Figure 3-11.

Figure 3-11 Comparison of the NPV of changes in soil carbon stocks at a national level from 2010 to 2060



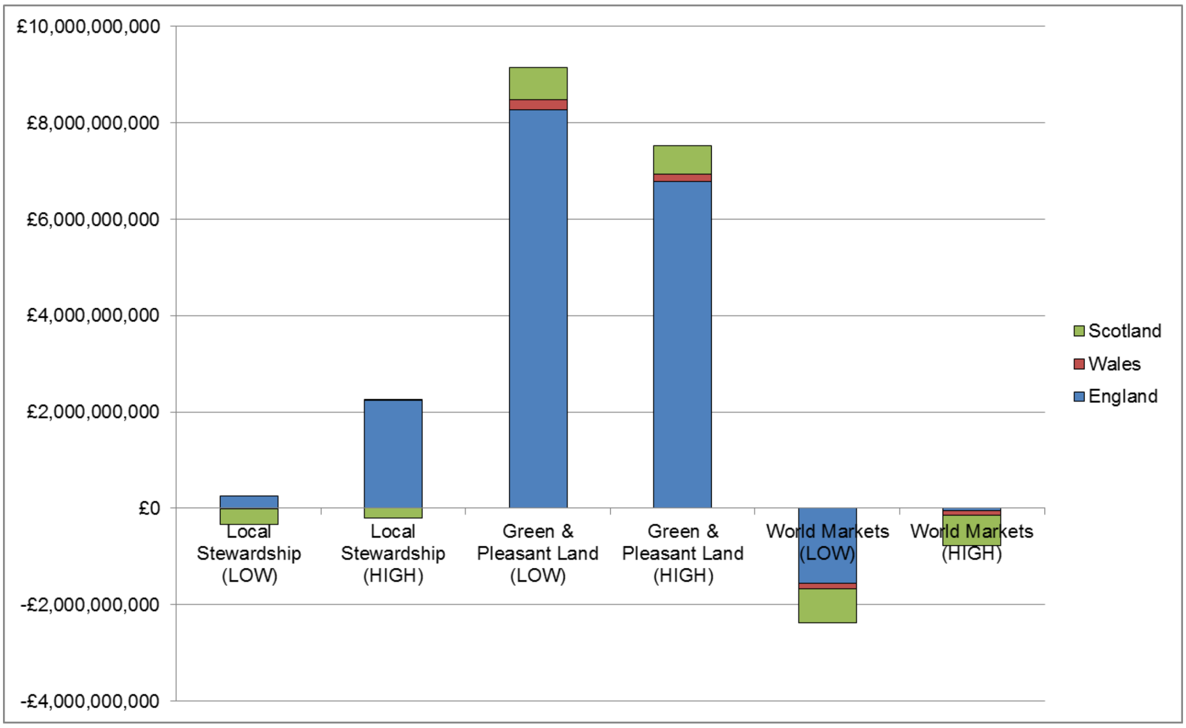
3.1.80 With regards to vegetation carbon stocks, the total value of the change across Great Britain over the period 2010 to 2060 ranges from a low of -£2.7 billion in the world markets (low emissions) scenario, to a high of £9.7 billion in the green and pleasant land (low emissions scenario). A comparison of the value estimates is presented in Figure 3-12, all values are presented in 2014 prices.

Figure 3-12 Comparison of the NPV of changes in vegetation carbon stocks across Great Britain from 2010 to 2060



3.1.81 Breaking down the estimates at a national level reveals an greater range of variation as set out in Figure 3-13.

Figure 3-13 Comparison of the NPV of changes in vegetation carbon stocks at a national level from 2010 to 2060



Priorities for further research

3.1.82 The findings presented in both the introductory material, and in our subsequent quantitative analyses, highlight a number of priorities for research on the UK's carbon stocks. Although all our recommended priorities focus on the collection or management of data, equally as important is the prompt and wide distribution of these data, so that researchers in agencies of Government, NGOs, academia and the general public can contribute towards its analysis and QA. We proceed by detailing each priority in turn.

1 How do soil and vegetation carbon interact?

3.1.83 Forest Research (2012) note that "there are very few complete C balances for UK woodlands", and only three long-term datasets on the CO_2 take-up by forests. Thus we know little about the passage of carbon through even the most studied vegetation ecosystems, and almost nothing about carbon budgets in unmanaged habitats, i.e. in unmanaged woodlands or scrub. Given that less intervention results in more total carbon stock in the vegetation (Forest Research 2012), unmanaged woodlands or scrub could be an important sink for CO_2.

3.1.84 Our analysis indicated that the potential expansion of semi-natural grasslands could be an important factor in future carbon balances of both vegetation and soil carbon. Yet this land cover type itself covers a huge array of taxa, communities and habitat structures. If we are to capture such soil-vegetation interactions in flux experiments, many of the habitat types are at present too broad to generalise across. Improving the skill in identifying biologically meaningful groups via remote sensing should also therefore be a priority.

3.1.85 Estimating carbon fluxes is difficult, as the large confidence intervals in IPCC (2013) demonstrate, but until monitoring experiments are conducted in a variety of contexts (and preferably over the longer-term), they will remain a largely unquantified uncertainty in our national carbon accounting. The CEH Carbon Catchments are an important example of the research benefits that can be derived from such monitoring.

2 What are the management techniques that can remove carbon from the atmosphere?

3.1.86 Land management has a large potential impact on soil carbon content, and could greatly alter carbon budgets. Based on the maximum amount of carbon in each cultivated soil type, Lilly and Baggaley (2013) estimated that Scottish soils have the potential to sequester between 150 and 215 MtC, should their management be optimised. This is roughly double our estimated changes for 'Local stewardship' and 'Green and pleasant land'. Both studies suggest that there is potential for land management to remove carbon from the atmosphere.

3.1.87 However, the techniques required to achieve this goal are still relatively untested, and where evidence does exist, the findings can be unclear (Thomson et al. 2012). For example, reducing or stopping tillage (cutting or ploughing of the soil) has been proposed as a means of reducing carbon emissions from agriculture. No-till treatments in combination with residue retention and crop rotation were even shown to significantly increase crop yields in dry climates (Pittelkow et al. 2015). It has also been shown to have beneficial effects on soil quality and water retention that make it a useful for climate change adaptation (Derpsch et al. 2010). Yet long-term monitoring of gas exchange has consistently failed to detect a carbon benefit of no-till (Baker et al. 2007, Powlson et al. 2014), and the effect of such an approach on nutrient retention following fertiliser application, and specifically not tilling the land during this process, is undocumented. It is likely that the technique is still promoted as an approach to mitigation (e.g. UNEP 2013), despite its mild effect on carbon sequestration, because (like many agricultural mitigation techniques) it has proven effective in some areas and not others (for a review see Smith et al. 2008). It also has beneficial effects on soil quality and water retention that will make it a useful for climate change adaptation. Thus the UK will need to establish the techniques that sequester carbon most effectively in its own national context, ideally via experimentation and monitoring. This will be especially pertinent for methane and nitrous oxide, where agriculture is responsible for the majority of the emissions.

3.1.88 The UK's forests contain an estimated 213 MtC, yet we know little of the feedbacks involved between the soil and the vegetation above, particularly for newly afforested areas (Forest Research 2012). Clearly, release of soil carbon will be more likely for planting techniques (e.g. earth movement, tillage) or species that disturb the soil more, and particularly so where these techniques require artificial drainage, which lowers the water table and enhance respiration. Beyond this, there is little information on the specific approaches that can be taken to minimise soil carbon release when planting. An experiment which examined the effects of various afforestation techniques on: a) the status of existing carbon in the soil; b) the uptake of soil and atmospheric carbon by the newly planted trees; and c) the outward transport of carbon via soil erosion/mobilisation of DOC; while also quantifying d) the emissions of associated fossil-fuel burning equipment, would allow the full carbon impact of tree-planting to be assessed. A recent global meta-analysis suggested that the land use prior to afforestation is the strongest control on its net carbon impact (Laganière et al. 2010), so experiments to assess which cover types (if any) sequester the most carbon when planted with trees would also be important.

3 To what extent are soil erosion models sensitive to future changes in extreme rainfall?

3.1.89 Although a substantial amount of progress has been made in reducing the uncertainty in projections of the more typical climate variables (e.g. mean temperature), it is apparent from the literature that current sets of climate projections (in which the UK is world-leading) do not offer researchers the potential to quantify the risks involved from the potential effects of 'less typical' climate variables, such as extreme rainfall events. It is possible that, where effects from the more 'typical' climate variables (e.g. mean temperature) are found to be less harmful (or even beneficial), actions to protect that asset are less likely to be prioritised. However, in reality the asset could face an unquantified, yet potentially substantial risk from 'less typical' climate variables. For example, a survey conducted in the aftermath of Storm St Jude showed that it had affected mature trees (high in C) disproportionately, caused damage at most of the survey sites visited, and resulted in an estimated 10 million trees being windthrown or snapped (Forestry Commission 2014b). Similar to the European drought of 2003, it is these extremes that could define where and how climate change affects us the most.

3.1.90 The next set of UK Climate Projections (UKCPNext) being developed by the UK Met Office are likely to have an increased focus on extreme events (alongside an emphasis on spatial coherency and near-term prediction). As we highlight above, it is not just the potential change in the magnitude of these events, but the change in their probability that is important, and improvements in the attribution of recent extremes (e.g. Christidis et al. 2015) should translate into improvements in the confidence in estimates of their future incidence.

4 How will the UK monitor and assess change in land cover?

3.1.91 Remote-sensing using satellites such as AVHRR and LANDSAT will undoubtedly continue to play a role, but rather than the intermittency of land cover mapping as is currently the case, for carbon accounting purposes it would be useful to improve the frequency of these measurements. To make an LCM 2007 tile, both a summer tile and a winter tile were required, to improve the contrast between cover types (Morton et al. 2011). It is therefore a possibility (should computing, satellites and resource allow) that the tiles could be updated on an annual basis (ONS 2015). To quality assure these data, field validation effort would have to increase, and the error associated with mapping the land cover would have to be substantially lower than the size of the changes in cover it would be required to detect. But it remains the case that a change in land cover is often cited in many environmental analyses, yet is rarely captured by the datasets being analysed. Such an annualised dataset would facilitate large advances in our understanding of many of the issues of environmental change, but the annualised datasets should use consistent methodologies to ensure they can be compared.

Conclusions

3.1.92 In this chapter, we have demonstrated how three different socio-economic directions result in substantially different outcomes for the soil and vegetation carbon stock of Britain. In the case of soil carbon stock, potential gains ranged from +6% to +11% (across two scenarios: 'Local stewardship', and 'Green and pleasant land'), while potential losses ranged from –12% to –13% (under 'World markets'). In the case of vegetation carbon stock, scenario uncertainty was even higher, ranging from +23% to +28% gains ('Green and pleasant land') to –3% to –8% losses ('World Markets'). The indirect effects of climate change were more pronounced on vegetation carbon stocks than on soil carbon stocks, heavily modifying the estimated net changes in vegetation carbon under 'Local stewardship' and 'World Markets'. In the latter scenario, the indirect effect on arable land was on a par with the effect of land cover change itself.

3.1.93 These headline results mask further spatial variation in the changes to stock we estimated, with land cover changes and climate change affecting the constituent countries of the UK to varying degrees (Table 3-20). Because England in particular was subject to a substantial number of drivers acting simultaneously (urbanisation, climate change, agricultural changes), uncertainties generated by differing socio-economic and climate futures resulted in larger variations in estimated carbon amounts. Wales and Scotland were subject to less pressure from these drivers, and thus estimates for these countries were less variable. These findings highlight the importance of applying a spatial approach in national assessments of natural assets.

Table 3-20 Range of potential equilibrium changes (± %) to carbon stock associated with the NEA scenarios for land cover change, by geographic area

Geographic area	Equilibrium changes in carbon stock driven by NEA scenarios for land cover change	
	Soil carbon	Vegetation carbon
Britain	–13% to +11%	–8% to +28%
England	–9% to +14%	–9% to +49%
Scotland	–8% to +3%	–5% to +5%
Wales	–12% to +7%	–3% to +6%

3.1.94 It must be noted that all our estimates represent equilibrium changes, and would in reality take anything from 50 to 750 years to occur (UK NIR 2014). This has important implications for the provision of an equable climate this century. In the case of vegetation carbon, the means of managing forests for carbon are becoming clearer (Forest Research 2012), and thus woodlands could play an important role in the UK's provision of an equable climate. The evidence for the efficacy of techniques to sequester carbon in the soil is more equivocal (Thomson et al. 2012), but because the potential gains are substantial, research will continue to focus on how these gains might be achieved. As ever, it will be necessary to balance any ambition to increase carbon stock with existing uses of the land (e.g. farming), where priorities can sometimes differ. Attention will therefore focus on approaches that bring multiple benefits (e.g. restoring upland peats for carbon, water and wildlife).

Wildlife

4 Wildlife

Introduction

4.1.1 Pole-ward species range shifts are an increasingly reported phenomenon, and likely caused by changing climatic conditions. These range shifts are particularly frequent for mobile species such as insects and birds and those with distribution limits associated with temperature such as alpine plants (Hickling et al. 2005; Hughes 2000). For example, both the Stag beetle (*Lucanus servus*) (Figure 4-1) and European bee eater (*Merops apiaster*) (Figure 4-2) are predicted to shift northwards as a result of a changing climate (BRANCH Partnership 2007).

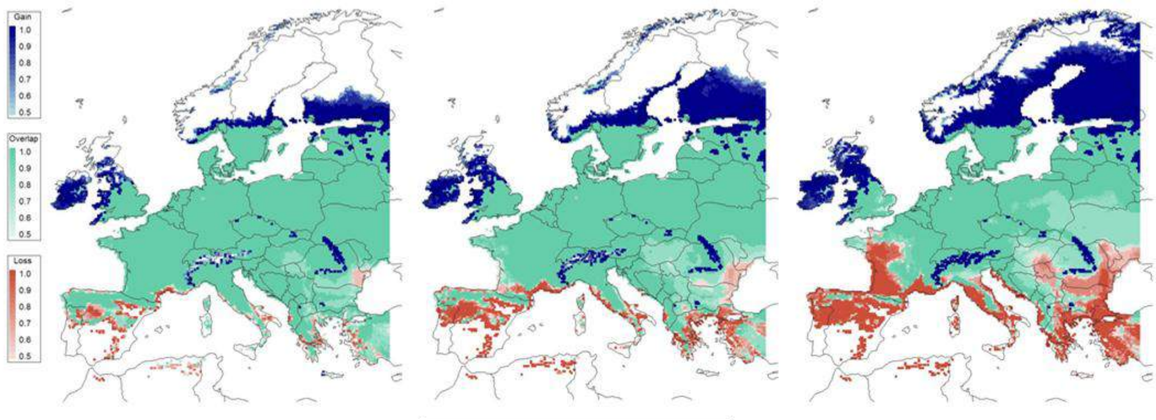


Figure 4-1 Projected range shift for the stag beetle (*Lucanus servus*) from the BRANCH project for the years 2020, 2050 and 2080 under the high climate scenario (HadCM3 A2).

(Source: BRANCH Partnership 2007.)

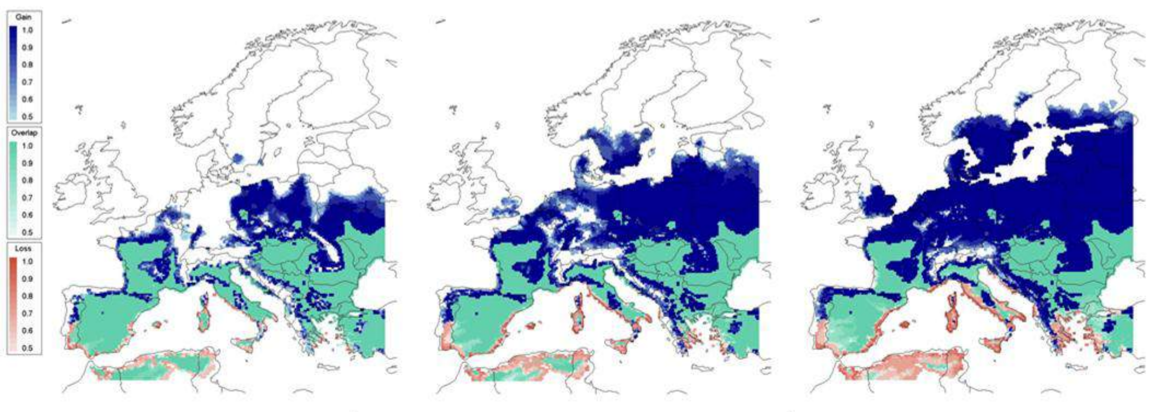


Figure 4-2 Projected range shift for the European bee eater (*Merops apiaster*) from the BRANCH project for the years 2020, 2050 and 2080 under the high climate scenario (HadCM3 A2).

(Source: BRANCH Partnership 2007.)

4.1.2 These changing distributions could have a significant impact on species protection, particularly because many projections suggest a change in species composition within existing protected areas. In order to target conservation resources efficiently and effectively, the robust and justifiable prioritisation of species for conservation action is needed. This should include a consideration of the likely vulnerability of species taxa to climate change, balanced against the potential opportunity that climate change may provide to other species.

4.1.3 This section considers an assessment of risks and opportunities that native species' in Great Britain and Northern Ireland will face as a result of climate change based on combining the outputs of distribution models that project future changes in species' distribution in response to climate change. Due to data constraints, this study does not take into account species colonising from Europe. Building on an earlier assessment of 3000 species for England (Pearce-Higgins et al. 2015), we extend the analysis to 4231 species across three climate scenarios, therefore making a very broad spatial assessment of the likely impact of climate change on biodiversity across Great Britain and Northern Ireland.

Approach and data

Methods and approach summary

4.1.4 Gridded distribution data at a 10km resolution were obtained for 4743 species across 17 taxonomic groups from the Biological Records Centre and the British Trust for Ornithology. To assess the impacts of changing climate (measures of summer and winter temperature, seasonality and moisture availability) on these species, we built state-of-the-art statistical models of the distribution of each species. Whilst conceptually similar to the modelling methods in previous reports such as the MONACH project (Harrison et al. 2001), these models differ in a number of fundamental ways. Firstly, they attempt to account for incomplete coverage by observers: many datasets are patchy, containing only records of known presence; gaps in records may reflect species absence or may simply mean no-one has searched for the species in the area. By using an occupancy model combined with taxa-specific estimates of observer effort, we can compute (with associated uncertainty) the probability of presence independently of the probability of observation. Secondly, our methods allow the quantification of the effects of unknown variables on the distribution in question: until recently all modelling methods assumed that distribution was a function of the climatic (or other) variables included in the analysis, plus some random error – in other words, all deterministic components of distribution were assumed by the modelling process to be known.

4.1.5 By contrast, our models assume that distribution is a function of both known climate covariates and spatially structured, but unknown factors. These may represent, for example, land use variables, additional climatic features not measured, patterns of anthropogenic impact. Thus, for the first time, we are able to quantify the relative effect of the climate variables we expect to be important, and other unknown variables. Thirdly, our method is spatially explicit and has been shown to accurately identify true statistical significance – a test failed by older methods. Finally, our method propagates uncertainties – in species distribution, in parameter estimation and in model output – exposing quite how uncertain many individual species projections are (which is why we do not provide individual level risks here), but nevertheless providing robust evidence for the overall patterns described below. Full details of the methods area available in Beale at al. 2013.

Data sources

Wildlife Data

4.1.6 Species distribution data for Great Britain and Northern Ireland were acquired from the Biological Records Centre and the British Trust for Ornithology at a standard 10 km resolution. These data contain records of where and when species have been observed. Within each group, the species selected were present in England and recorded on more than five squares of 10 x 10km in Great Britain to ensure there were a minimum amount of data for modelling (Hickling et al. 2006). Data were available for the following groups: ants, spiders, bees, birds, bryophytes, butterflies, carabid beetles (ground beetles), centipedes and millipedes, cerambycid beetles (long-horned beetles), coccinelids (ladybirds), craneflies, moths, dragonflies and damselflies, plants, soldier beetles and wasps. Data for hoverflies and crickets and grasshoppers were not available for Northern Ireland and were therefore not included in the analysis. Cells for which climate data were not available, because they were partially marine cells, were also excluded from analyses. Cells on small islands, isolated from the UK mainland, were also excluded to aid model convergence. This gave 2826 10 km × 10 km cells for inclusion in analyses of both Biological Records Centre and British Trust for Ornithology data. In total, this yielded 4743 species. However, for some very sparsely recorded species, models failed to converge (see below), giving a total of 4231 species for which distribution models were produced (Table 4-1).

4.1.7 To determine species distribution, we used distribution data from the period 1970-89. In addition to providing a baseline for distributions, this time period was used because, with an increasing magnitude of climate change being recorded after this period, more recent distributions may be increasingly out-of-step with the climate. Native vascular plants and birds were slight exceptions. For plants, in order to be consistent with the start and end date of major atlases, we used the period 1970-86. For birds, the first survey period spanned 1988-91, again for consistency with a breeding bird atlas (Gibbons et al. 1993).

4.1.8 For birds and plant species distributions, where available, data from both Great Britain and Northern Ireland and Europe were used in the modelling process described below. Using the occurrence data from European distributions may help to improve the accuracy of modelling UK species as it typically contains information on distribution limits (northern or southern) that are not currently present in the UK. Further details are provided in the statistical modelling section. Distribution data were not available in Europe for other species groups possibly making the results for these species somewhat less reliable, although we have previously shown that for plant and bird species for which European data were available, models built either with or without Europe had remarkably little influence on UK projections giving a degree of confidence to our results for all species with data available only in the UK. On the other hand, a lack of European data precluded widespread modelling of potential new colonists with no current UK distribution, which necessarily means the results below lack many likely colonists in the south.

4.1.9 Species distribution data for Europe were acquired from the European Bird Census Council and the Atlas Florae Europapeae (AFE). For birds, distributions at the European and British scales were identified by matching species names in the two datasets. For plants, however, taxonomy sometimes differed between the two datasets, so distributions of each species were identified in two stages. First, species with identical names in both datasets were identified. Second, if the genus was present in the AFE data but the species name was unmatched, species were manually matched by searching for synonyms or subspecies in the European dataset. For any British species with multiple matches (e.g. when multiple subspecies were listed in the European data but only one was identified in UK), the European data were combined into a single distribution. Within the European data, presences were defined only by records of native occurrences. For all European datasets, cells from Eastern Europe were not included to avoid problems of low observer effort; the maximum longitude was 29.99°. Iceland and the Faroe Islands were further excluded from model fitting to aid model convergence. This gave 2,644 50 km × 50 km cells for inclusion in analyses.

Climate data

4.1.10 For models at the UK scale, observed climate data, on a 5km × 5km grid, from the period 1961-90 were downloaded from the UK Met Office.[11] These were taken to represent the baseline climate that would be used to describe observed species distributions. Climate change data were downloaded from the UKCP09 user interface.[12] To ensure that climate data were consistent across adjacent grid cells and to ensure that different climate variables were consistent within the same grid cell, it was decided that the Spatially Coherent Projections were the most suitable of the UKCP09 products. The alternative probabilistic projections that are conditioned on global temperature changes are not spatially consistent between cells (hence should not be used for spatially explicit modelling) and are not consistent between variables within the same cell (hence

[11] See http://www.metoffice.gov.uk/climatechange/science/monitoring/ukcp09/
[12] See http://ukclimateprojections-ui.defra.gov.uk

multiple climate variables from a given cell should not be used simultaneously). To represent UK climate under global temperature changes of 2°C, 4°C and 6°C within the Spatially Coherent Projections, we identified the time period and SRES scenario that were used to produce the global-temperature-change products: 2070-99 for scenario B1 (2°C change), 2070-99 for scenario A1B (4°C change) and 2070-99 for scenario A1B (6°C change).[13] As the Spatially Coherent Projections have data from 11 RCM ensemble members, to which no probability or certainty is attached, all 11 ensemble members were used to generate projections, giving a total of 33 future climate datasets.

4.1.11 We extracted mean temperature (°C), cloud cover (%) and total rainfall (mm) on a monthly timescale. Observed data were aggregated to a 10 km × 10 km grid using the mean value. Climate change data were provided at an approximate 25 km resolution, but represented a 25 km resolution change value. Change values were applied to the underlying 5 km resolution observed data. Averages of the resulting surfaces were then calculated at 10 km resolution. The resultant maps gave absolute values for each climate variable in the future scenarios at the required resolution.

4.1.12 For European-scale models, observed climate data from the period 1961-90 were acquired from the Tyndall Centre for Climate Change Research; dataset CRU TS 1.2.4[14] These data were aggregated using means to the 50 km UTM grid as used in the AFE. As with UK climate data, we used mean temperature, cloud cover and total rainfall on a monthly timescale.

4.1.13 For both UK and European analyses, raw climate data were converted into appropriate bioclimatic variables that should have more direct influences on species' distributions. These were:

- mean temperature of the coldest month (hereafter MTCO): a measure of winter cold
- growing degree days (hereafter GDD5): a measure of summer warmth
- the coefficient of variation of temperature (hereafter cvTemp): a measure of seasonality
- soil moisture (hereafter soilWater): a measure of moisture availability

4.1.14 These variables have been shown to correlate strongly with many species distributions in a variety of species groups, from plants to birds and butterflies (Araújo and Luoto 2007; Araujo et al. 2005; Huntley et al. 2008).

4.1.15 Choice of variables was made to reflect two primary properties of climate – i.e., energy and water – that, on the basis of prior knowledge, have known roles in imposing constraints upon plant and butterfly species distributions as a result of widely shared physiological limitations (e.g. Prentice et al. 1992; Hill et al. 1999; Parmesan et al. 1999).

4.1.16 MTCO was calculated by simply finding the lowest monthly temperature for each cell. GDD5 was calculated by fitting a spline to mean monthly temperatures for each cell, and then summing the number of days with temperatures greater than or equal to 5°C. This is a standard threshold and reflects the general properties of photosynthesis and plant enzymes resulting in little plant growth with temperatures below 5°C. cvTemp was calculated by converting mean monthly temperatures to °K, and then dividing the standard deviation by the mean for each cell. Finally, soilWater was calculated following the bucket model described by Prentice et al. (1993), which takes inputs of temperature, rainfall, percentage sun/cloud and soil water capacities then calculates the soil water balance over the year for each cell. This method takes into account evapotranspiration.

[13] See http://ukclimateprojections.defra.gov.uk/22614
[14] See http://www.cru.uea.ac.uk/~timm/grid/CRU_TS_1_2.html

Statistical modelling

4.1.17 We applied a Bayesian, spatially explicit (Conditional Autoregressive) Generalised Additive Occupancy Model to species' distribution data in order to separate climatic, spatial and random components in determining the distribution of each species (Beale et al. 2014). This approach addresses the problem of spatial autocorrelation in large-scale species' distribution data (Beale et al. 2008), and fits flexible relationships between species' occurrence and the climate data.

4.1.18 For those taxa for which European data were available, models were initially constructed using uninformative priors (i.e. we have no prior knowledge of what this relationship should be) across Europe to describe the relationship between occurrence and climate. Once converged, a second model was fitted to the finer-scale distribution data from the UK using informative priors from the European-scale analysis. As a result, any strong climatic signal based on the European distribution would remain essentially unchanged when modelled using UK data only. However, if there is uncertainty at the European-level in climate sensitivity, the UK-scale model would be more informed by local conditions.

4.1.19 In cases where there was uncertainty in the estimation of species' response at a European level, then the British model would be more heavily informed by outputs from the British component of the model. For species for which data from the UK only were available, only the second model was conducted but using uninformative priors rather than priors based upon information from the European distribution.

Methods for projecting species distributions

Extracting parameter estimates

4.1.20 From all models, we used 1000 parameter estimates for all subsequent analyses. The median, 2.5 percentile and 97.5 percentile of the 1000 estimates were extracted for each parameter (e.g. for each climate variable), including the intercept. The median estimate of each parameter was used to generate subsequent projections.

Producing probability estimates

4.1.21 To calculate the total probability of species' occurrence for a given cell, the bioclimatic variable values for that cell were multiplied by the parameter estimates, and the intercept and random effects values were added. The inverse logit (a function transforming the model outputs to probabilities) of these values was then calculated to give values on the probability scale. This was firstly done for the baseline climate period establishing current/recent climatic suitability and defining the current range. Following this, the same procedure was carried out using the future climate data. All 11 RCM ensemble members were used individually within each climate change scenario (i.e., +2°C, +4°C +6°C), meaning that there were 33 projections for each species.

Defining the current range

4.1.22 Following Beale et al 2014, we defined the current range to include all cells with observed presences, as well as cells which had an occurrence probability equal to or higher than any cells with observed presences.

Estimating range occupancy

4.1.23 Overall range occupancy (the number of cells expected to be occupied by the species) was estimated by summing the probabilities across cells; this was done separately for cells within the current range and cells outside the current range.

Identifying priority areas for conservation

4.1.24 Protected areas are a cornerstone of conservation policy in the UK, and an important part of identifying adaptation strategies for climate change is the identification of areas where biodiversity is currently, or is soon likely to be, underrepresented within the protected area network. Systematic conservation planning has been developed as a tool that allows spatial prioritisation based on species' (and other) distributions, aiming to identify the optimal arrangement of land protection to adequately protect as many species as possible. Various tools are available for this process, but one, the Zonation algorithm (Moilanen et al. 2005), is ideally suited to the modelled distributions we generated, as it accepts probabilities as input variables.

4.1.25 Zonation works by assuming all cells in the landscape are protected, and then seeks to find the cell that can be 'unprotected' with least impact on the overall biodiversity remaining within the network. This process is repeated until no cells are protected, providing an effective ranking of the importance of cells across the landscape. There are a myriad of ways in which total biodiversity value can be calculated, as well as the weightings given to individual species and the precise rules for cell removal. Our analysis here represents a first draft of the process, highlighting the broad patterns of protection. We assumed that the current protected area networks (consisting of Special Areas of Conservation (SACs), Special Protection Areas (SPAs) and Sites of Special Scientific Interest (SSSIs)) would remain in place, and used Zonation, on each network individually, to prioritise the remaining cells for additional protection under current conditions, and under the future scenarios. Furthermore, we assessed, for any given proportion of land protected, what proportion of species were 'adequately protected', for a variety of definitions of adequately protected based on the absolute number of cells estimated to be occupied falling within the remaining protected area.

Results summary

Model summaries

Table 4-1 Data sources and number of species included in the results across all taxonomic groups

Taxon	Recording scheme	Number of species for which climate models converged
Ants	Bees, Wasps and Ants Recording Society	34
Native vascular plants	Botanical Society of the British Isles	1410
Bryophytes	British Bryological Society	469
Moths	Butterfly Conservation	662
Spiders	British Arachnological Society, Spider Recording Scheme	456
Coleoptera-Carabids	Ground Beetle Recording Scheme	281
Bees	Bees, Wasps and Ants Recording Society	208
Wasps	Bees, Wasps and Ants Recording Society	175
Birds	British Trust for Ornithology	158

Centipedes and Millipedes	British Myriapod and Isopod Group, Centipede Recording Scheme	72
Diptera-Craneflies	Dipterists Forum, Cranefly Recording Scheme	74
Butterflies	Butterfly Conservation	58
Ladybirds	Ladybird Recording Scheme	38
Long-horned beetles	Cerambycidae Recording Scheme	44
Dragonflies/Damselflies	Dragonfly Recording Network	41
Soldier-beetles	Soldier Beetles, Jewel Beetles and Glow-worms Recording Scheme	51

Risks among taxonomic groups

4.1.26 Across all three climate scenario projections (low, 2^{o}C change; medium, 4^{o}C change; and high, 6^{o}C change), the taxonomic groups showed similar patterns of range change; with the exception of bryophytes, all groups, because of a positive value, were predicted to increase in spatial range (Figure 4-3). Bryophytes were the only group predicted to experience a decrease in climate space across the three climate scenarios. Although the mean proportional change in climate space available to plants, dragonflies, long-horned beetles and birds is expected to increase, these groups showed much less change than other groups.

4.1.27 Overall, change in climate space showed little variation across taxonomic groups for the three climate scenarios, with relative change increasing slightly from the low to the high scenario, and the greatest relative change taking place between the medium and high scenarios.

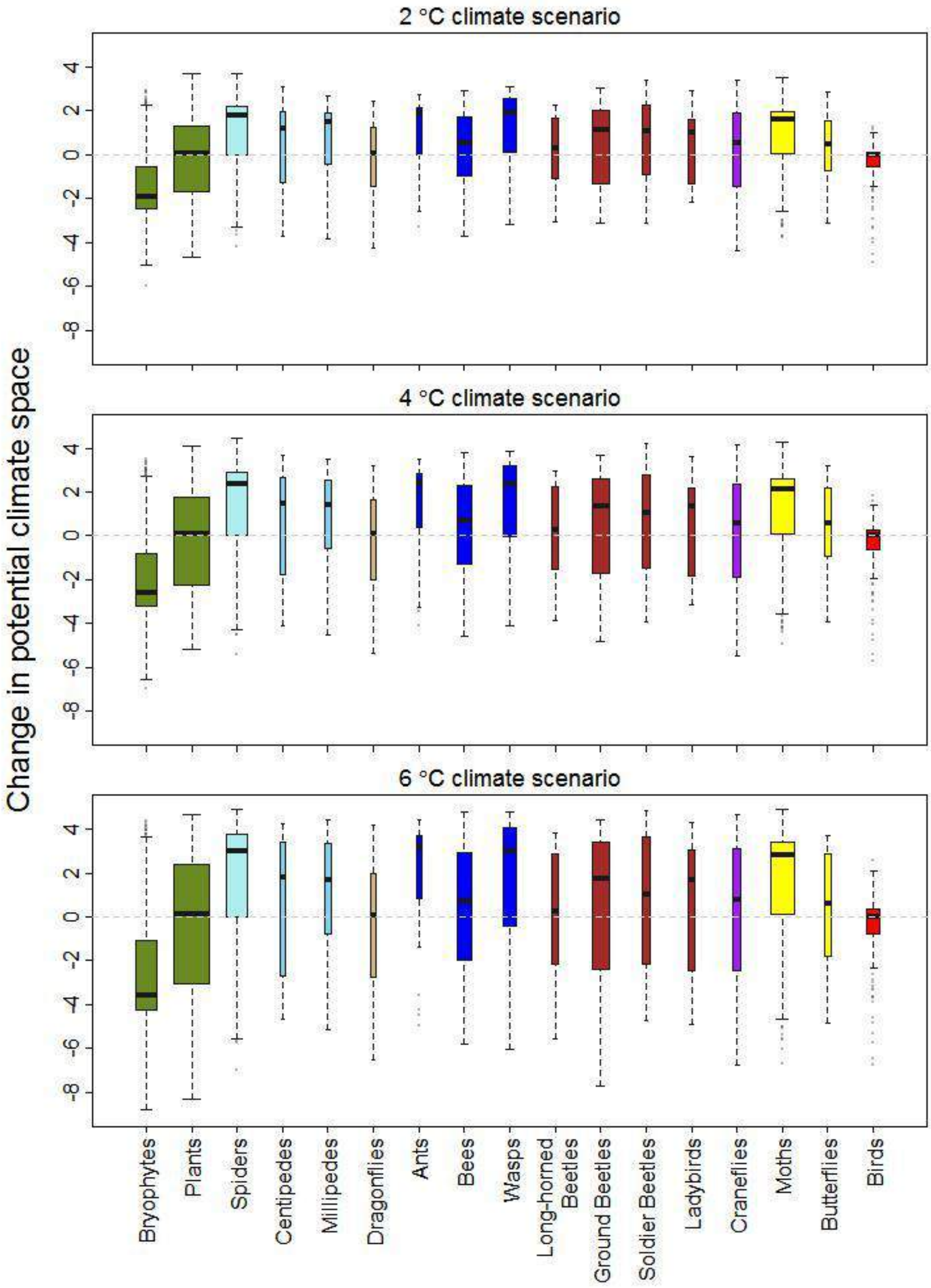


Figure 4-3 Overall forecasted range change as a proportion of possible change by taxonomic group under *2*°C , *4*°C *and 6*°C climate scenarios. Colours relate to different taxonomic groups and box widths are relative to number of species in each group.

(Change in climate space is presented on a logit scale, which is the log of the odds ratio).

4.1.28 Across all groups, taxa showed a similar sensitivity to climatic variables, suggesting that intrinsic vulnerability to climate change is similar across all groups, with impact dependent mostly upon exposure to actual climatic change rather than differences in sensitivity , i.e. change in potential climate space for each species group is likely to be similar across taxa (Figure 4-4).

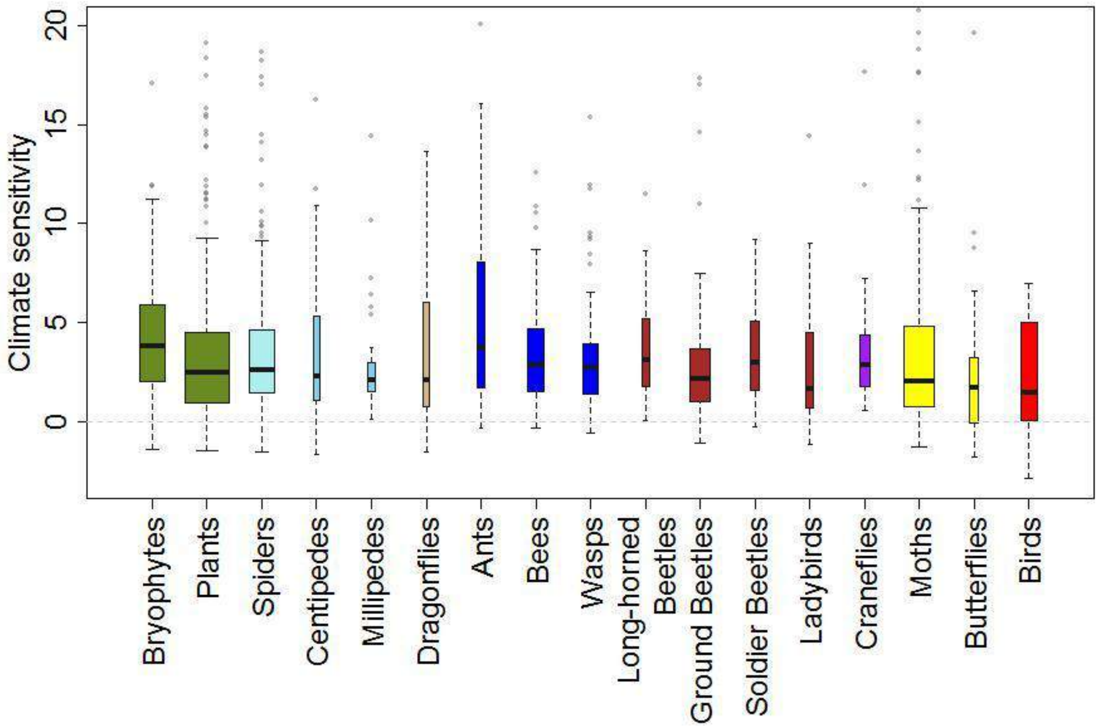


Figure 4-4 Climate sensitivity - proportion of modelled distribution pertaining to climate signals versus other processes by taxonomic group

(Change in climate sensitivity is presented on a logit scale, which is the log of the odds ratio).

Functional type results

4.1.29 Relatively more primary producers (organisms which are not dependent on others for food, e.g. plants generate food through photosynthesis) appear to be more negatively impacted than organisms in other trophic levels, presumably reflecting the more negative impacts on bryophytes (liverworts and mosses). This is of potential concern, as they are fundamental importance for ecological networks, but this should also be balanced against likely positive impacts of CO_2 fertilisation not considered here (Figure 4-5).

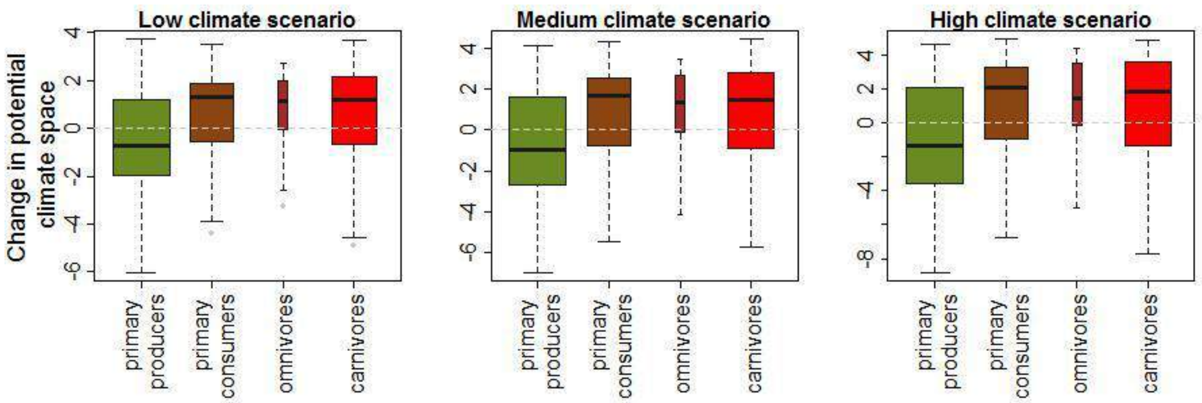


Figure 4-5 Expected proportional range change by trophic level. Box widths are relative to number of species in each trophic level.

(Position of organisms within an ecological food chain; herbivores consume primary producers, omnivores consume from different trophic levels and carnivores consume other animals).

Geographic risk

4.1.30 The geographic risk results presented here are risks across all modelled species combined.

Areas likely to see most local extinctions

4.1.31 Areas with proportionately more local species extinctions (i.e. species that are lost from a particular cell but may remain present in other cells across the UK) are focussed on northern areas and areas of higher topography (Figure 4-6). Areas in the south of the UK, particularly along coastal areas and around urban areas are predicted to have the lowest percentage of local species loss. The Western Isles, Shetland and Northern Ireland all have relatively low local extinction probabilities, perhaps reflecting the already depauperate nature of their flora and faunas. Across the three climate scenarios, the number of cells with high risk of local species loss increases.

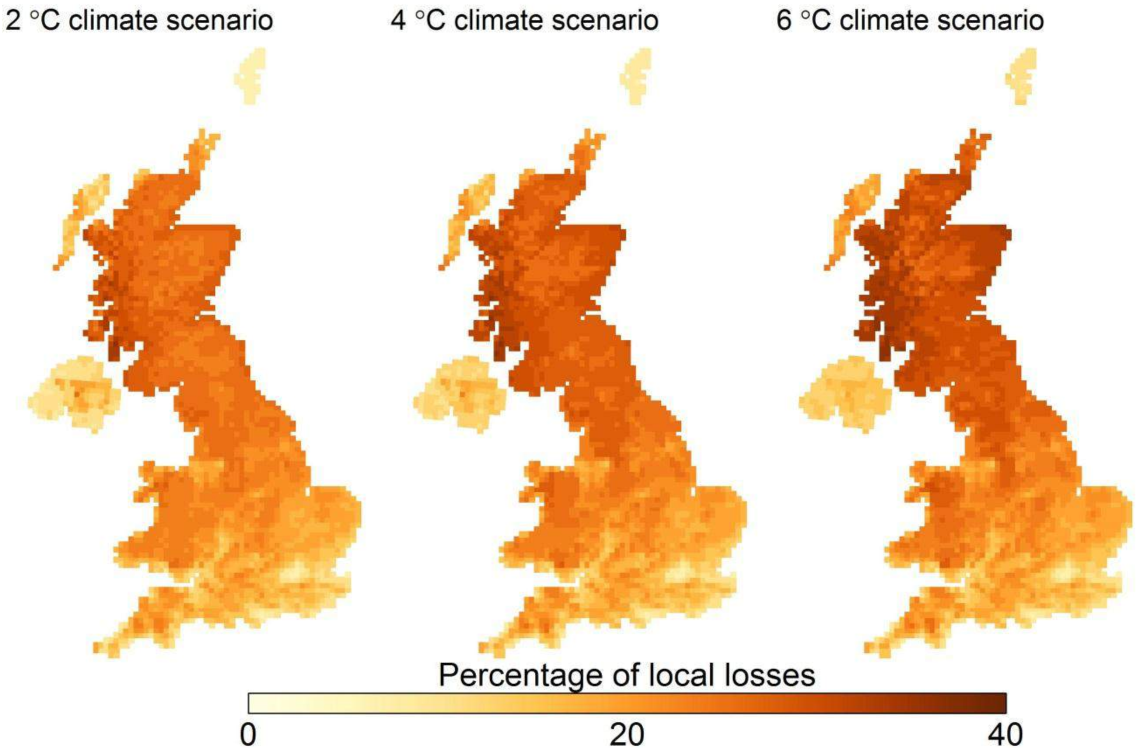


Figure 4-6 Areas likely to see local extinctions

(i.e. local loss of species) per 10km grid cell for 2°C , 4°C and 6°C climate scenarios across all taxonomic groups).

4.1.32 Because there is a strong species richness gradient in the UK (i.e. more species in the south than the north), the actual numbers of species likely to lose climate space from each square has a slightly different pattern to the percentage of species likely to lose climate space: compare Figure 4-6 and Figure 4-7. The greatest numbers of local species losses are expected in western Scotland, and western upland areas of England and Wales such as the Lake District and Snowdonia. Areas with lowest levels of species loss include the Western Isles, northern Scotland and South East England. The patterns of losses across the three climate scenarios are similar, but the number of areas with higher species loss increases from the low to the high climate scenario. The maximum and minimum numbers of species lost remains similar across the three climate scenarios.

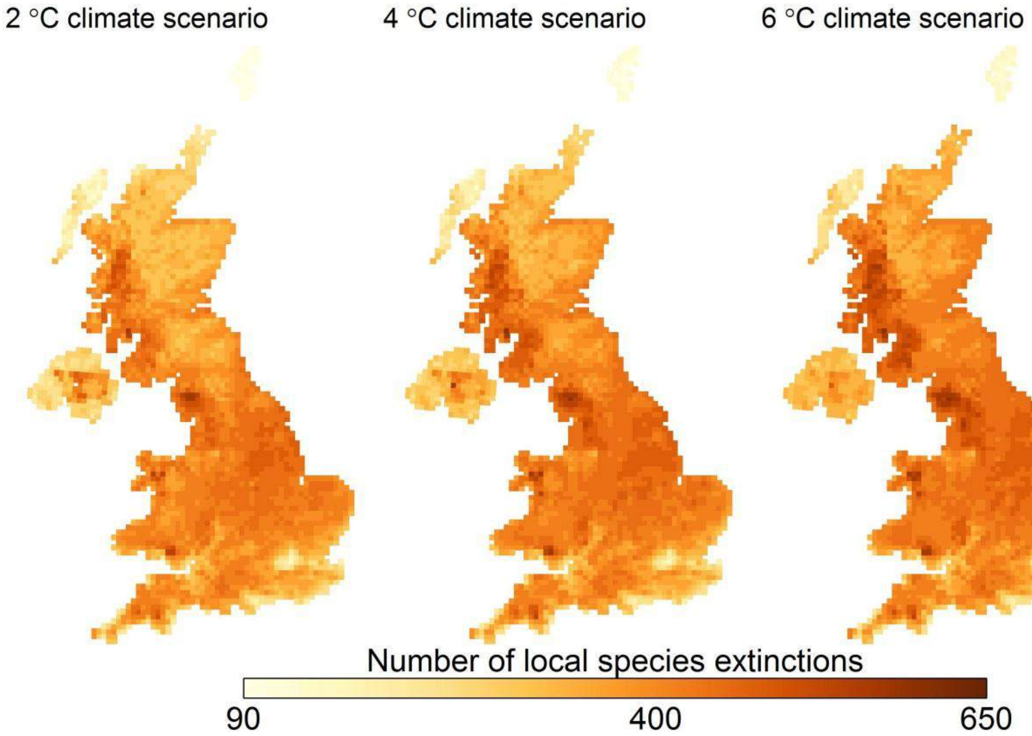


Figure 4-7 Numbers of local species loss

(i.e. loses per 10km grid cell for 2°C , 4°C and 6°C climate scenarios across all taxonomic groups.

4.1.33 Maps of uncertainty highlight that upland areas and islands have a lower uncertainty of local losses (Figure 4-8). Uncertainty also increases from low to the high climate scenarios. These maps suggest that the relatively lower percentage of species loss in the Western Isles, Shetland and Northern Ireland (Figure 4-6) is due to lower existing species diversity rather that high uncertainty.

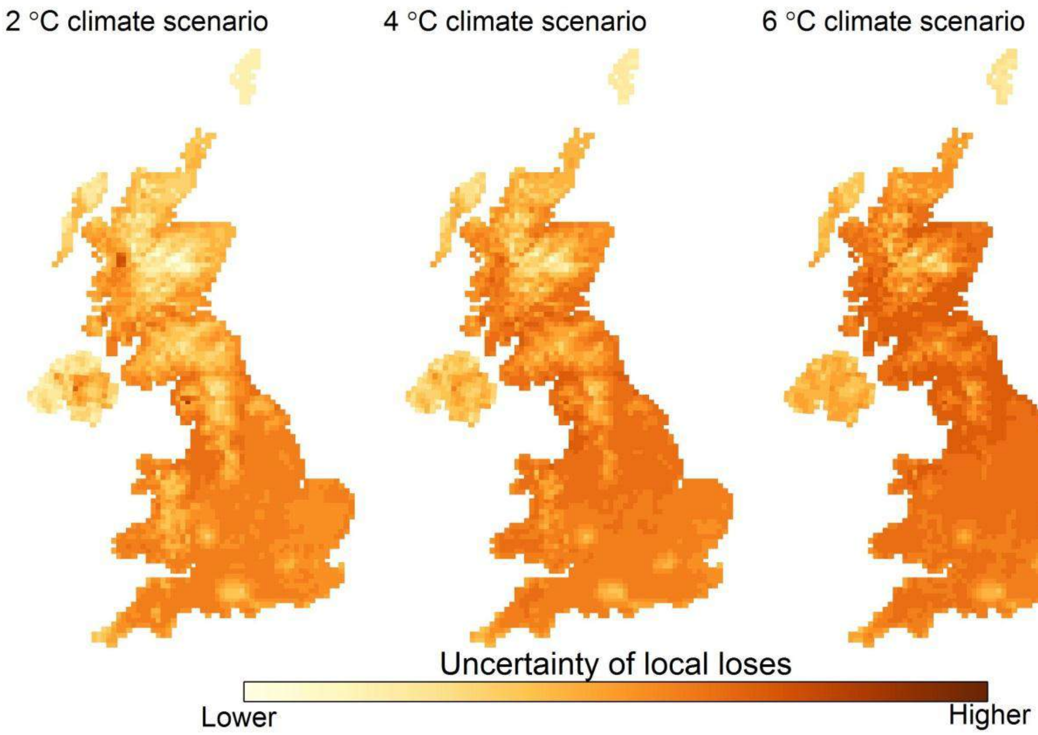


Figure 4-8 Uncertainty in areas likely to see local extinctions

(i.e. uncertainty of local species loss) per 10km grid cell for 2°C , 4°C and 6°C climate scenarios across all taxonomic groups.)

4.1.34 Separating the percentage of currently occupied cells for each species likely to experience local extinction by taxonomic group revealed significant variation in local losses between groups (Figure 4-9). Bryophytes and long-horned beetles show the greatest proportion of local losses per 10km grid cell and moths the lowest proportion of losses. These patterns are similar across the three climate scenarios.

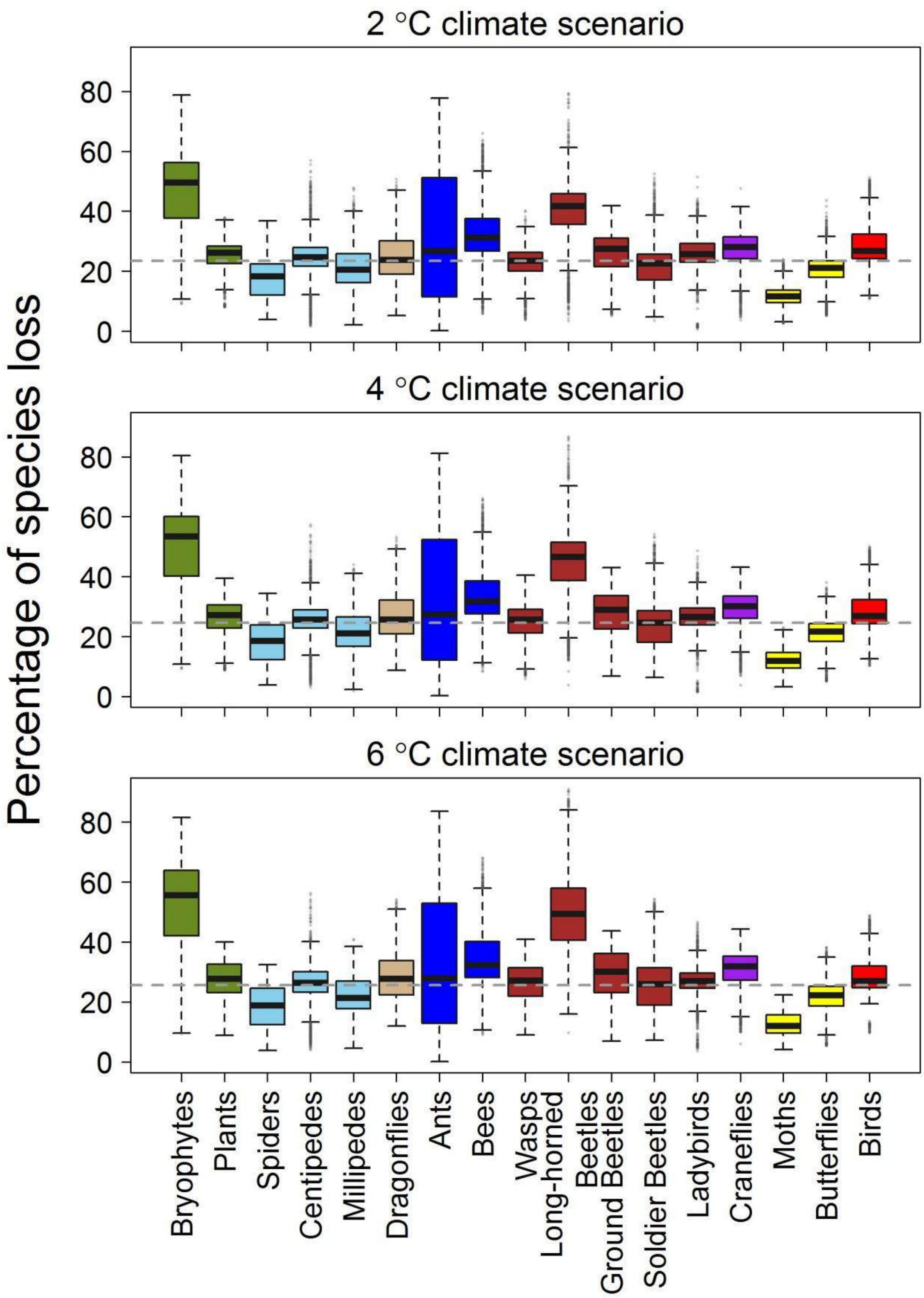


Figure 4-9 Percentage of local extinctions

(i.e. local loss of species), within 10km grid cells, per taxonomic group for 2°C , 4°C and 6°C climate scenarios. Horizontal dashed line represents the mean value across all species groups).

4.1.35 Among taxonomic groups, there was a wide variation in the level of local species loss and these patterns were consistent across the three climate scenarios (Figure 4-10). Bryophytes and plants were predicted to have some of the greatest numbers of losses in comparison to other groups, perhaps due to the high number of individual bryophyte and plant species, which are also are highly sensitive to changing climatic conditions.

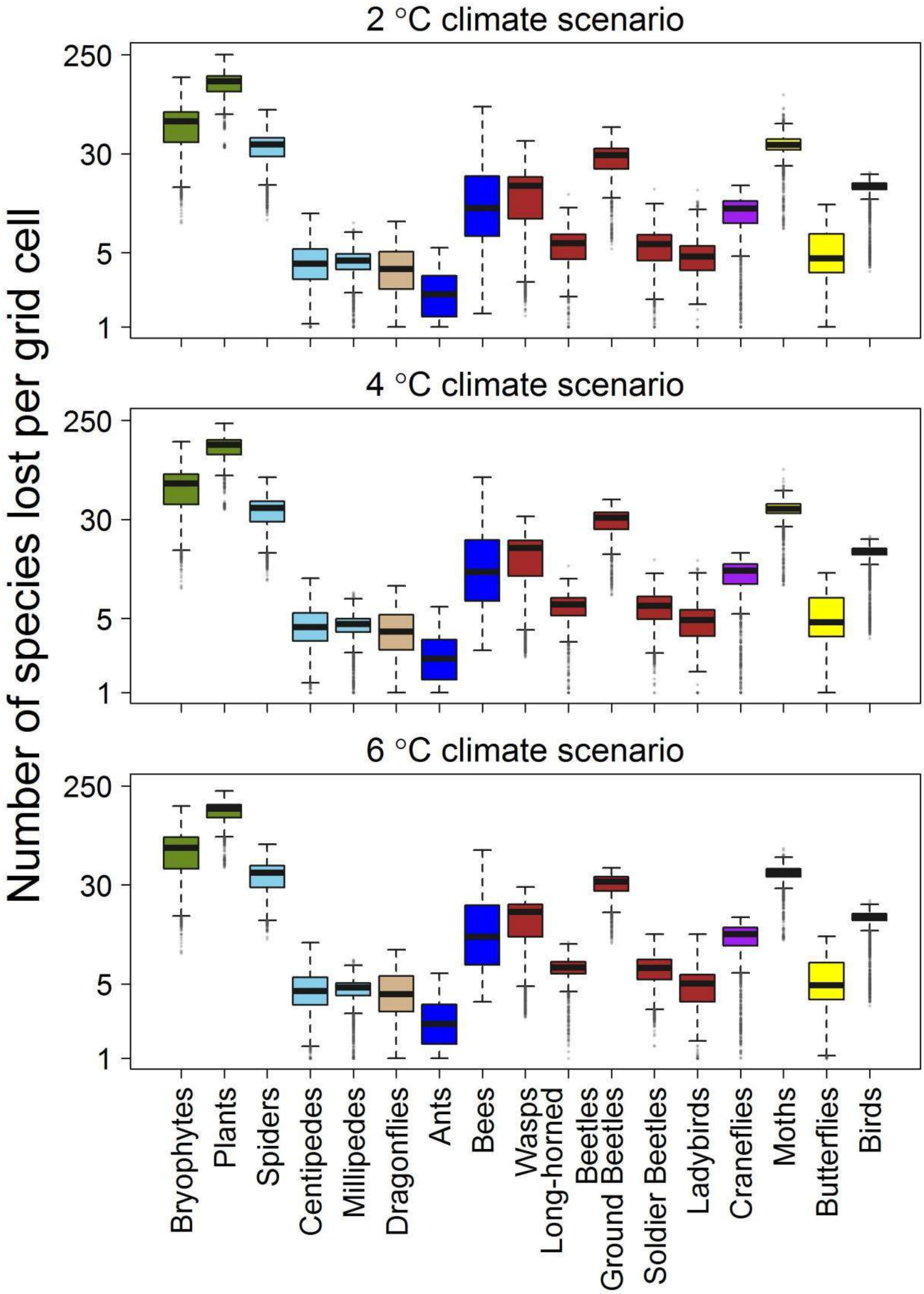


Figure 4-10 Number of local extinctions

(i.e. local loss of species, within 10km grid cells, per taxonomic group for 2°C , 4°C and 6°C climate scenarios (note logarithmic scale of y-axis).

Areas likely to see most colonisations

4.1.36 Similar spatial patterns to local extinctions occur for local colonisations. Colonisations are greatest in northern England and Scotland; however, in contrast to local extinctions, upland areas, particularly under low (2°C) and medium (4°C) climate scenarios have a relatively low colonisation probability (Figure 4-11). These maps represent the number of likely colonisations from existing species. It is important to remember that these results ignore potential colonists from further south, which are likely to be significant in number for many groups.

4.1.37 Areas in southern England, the Welsh coastline, Northern Ireland and parts of the Scottish highlands have a similar colonisation probability between the low and medium climate scenarios. With the exception of the Scottish highlands, this pattern stays the same under the high climate scenario, while there is also an increase in the probability of colonisation in upland areas of Wales and northern England under the high (6°C) climate scenario.

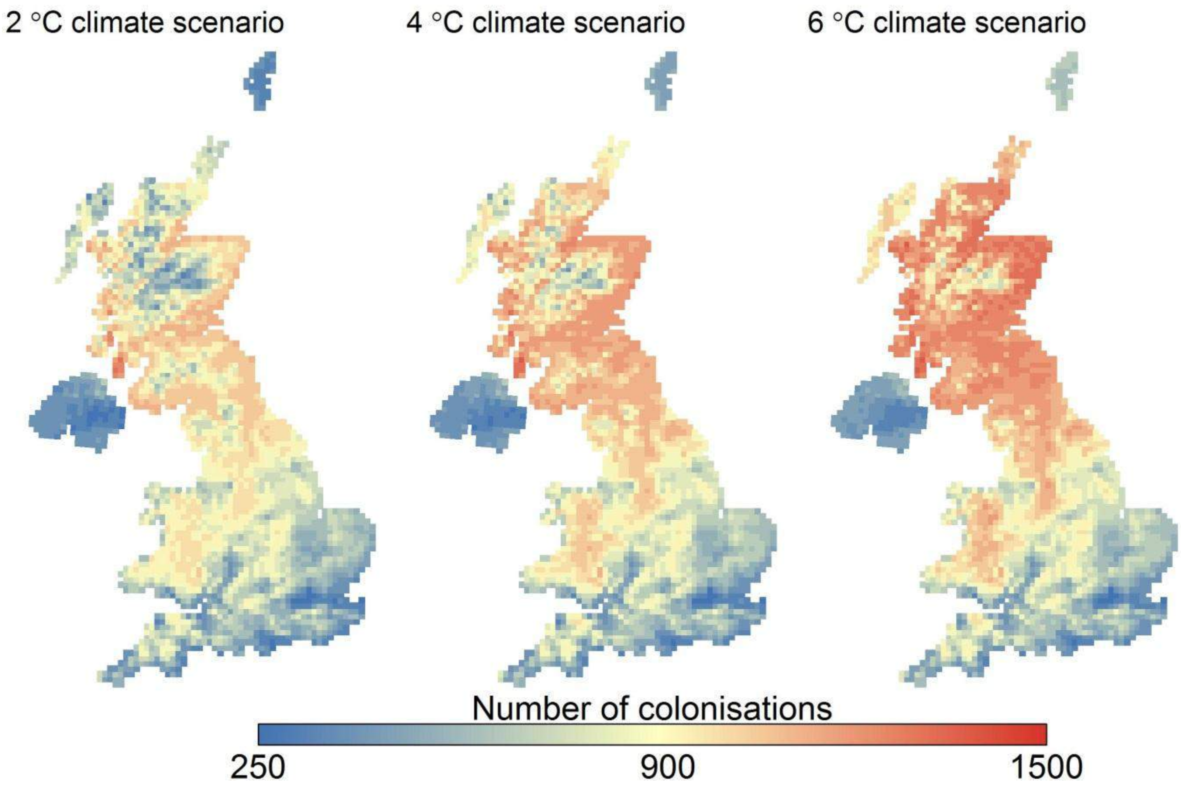


Figure 4-11 Areas likely to see colonisations from existing UK species

(per 10km grid cell for 2°C , 4°C and 6°C climate scenarios across all taxonomic groups)

4.1.38 Uncertainty of colonisations increases from the low to high climate scenarios, but the areas of low uncertainty, such as Northern Ireland, South West England and parts of the Lake District remain constant across the three scenarios (Figure 4-12).

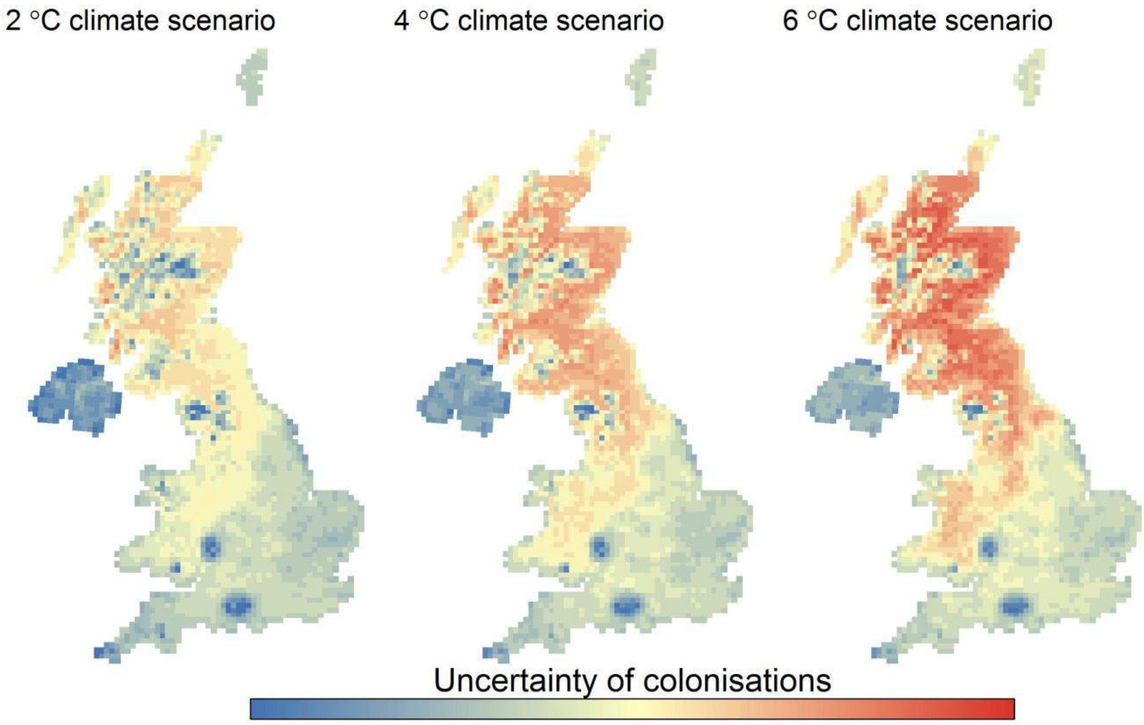


Figure 4-12 Uncertainty in areas likely to see colonisations from existing UK species

(per 10km grid cell for 2°C , 4°C and 6°C climate scenarios across all taxonomic groups.)

Areas likely to see greatest species turnover

4.1.39 With respect to the areas likely to see most species turnover, as is obvious from the above plots of colonisations and local extinctions, turnover is also expected to be highest in the north, where a considerable shift in communities can be expected. The Simpson Index (Figure 4-13) is a measure of the change in community, independent of any change in richness of that community, and is commonly used to illustrate spatial or temporal turnover (Koleff et al. 2003).

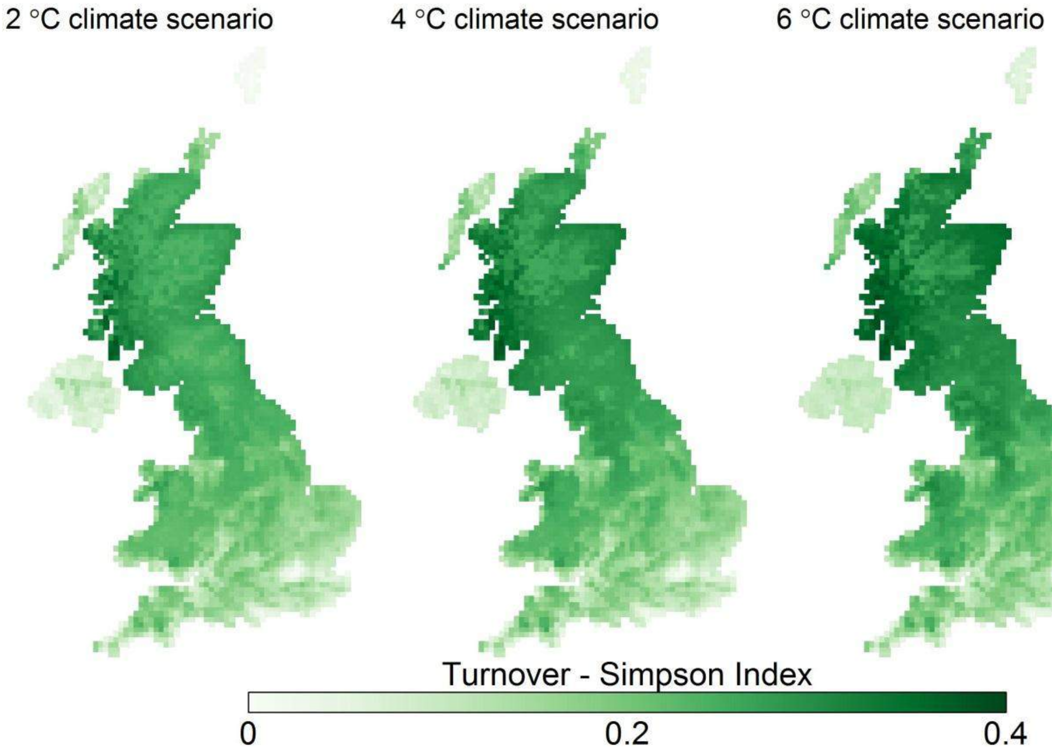


Figure 4-13 Areas likely to see species turnover (Simpson Index)

(per 10km grid cell for 2°C , 4°C and 6°C climate scenarios across all taxonomic groups.)

Impacts on protected areas

4.1.40 The probability of local extinctions and colonisations, across all taxonomic groups, inside and outside different protected areas, is shown in Figure 4-14. For both local extinctions and colonisations, there is a higher probability of change inside Special Areas of Conservation (SACs) Special Protection Areas (SPAs) and Sites of Special Scientific Interest (SSSIs) than outside, indicating greater species turnover in these areas than the surrounding regions. The greater change expected within protected areas than outside them is likely to reflect the general tendency for protected areas to be sited in regions of atypical and more marginal environmental conditions. Rare species often occur in such areas, but unusual environmental conditions are more likely to see change that impacts species presence than areas where conditions are more widespread and generalist species are more abundant.

4.1.41 Within protected areas, the proportion of local extinctions and relative colonisations (the number of colonisations divided by the original number of species per 10km grid cell) increases between the low and high climate scenarios. In comparison, outside protected areas, both colonisation and local extinction show a smaller increase in local colonisation and extinctions across the three climate scenarios.

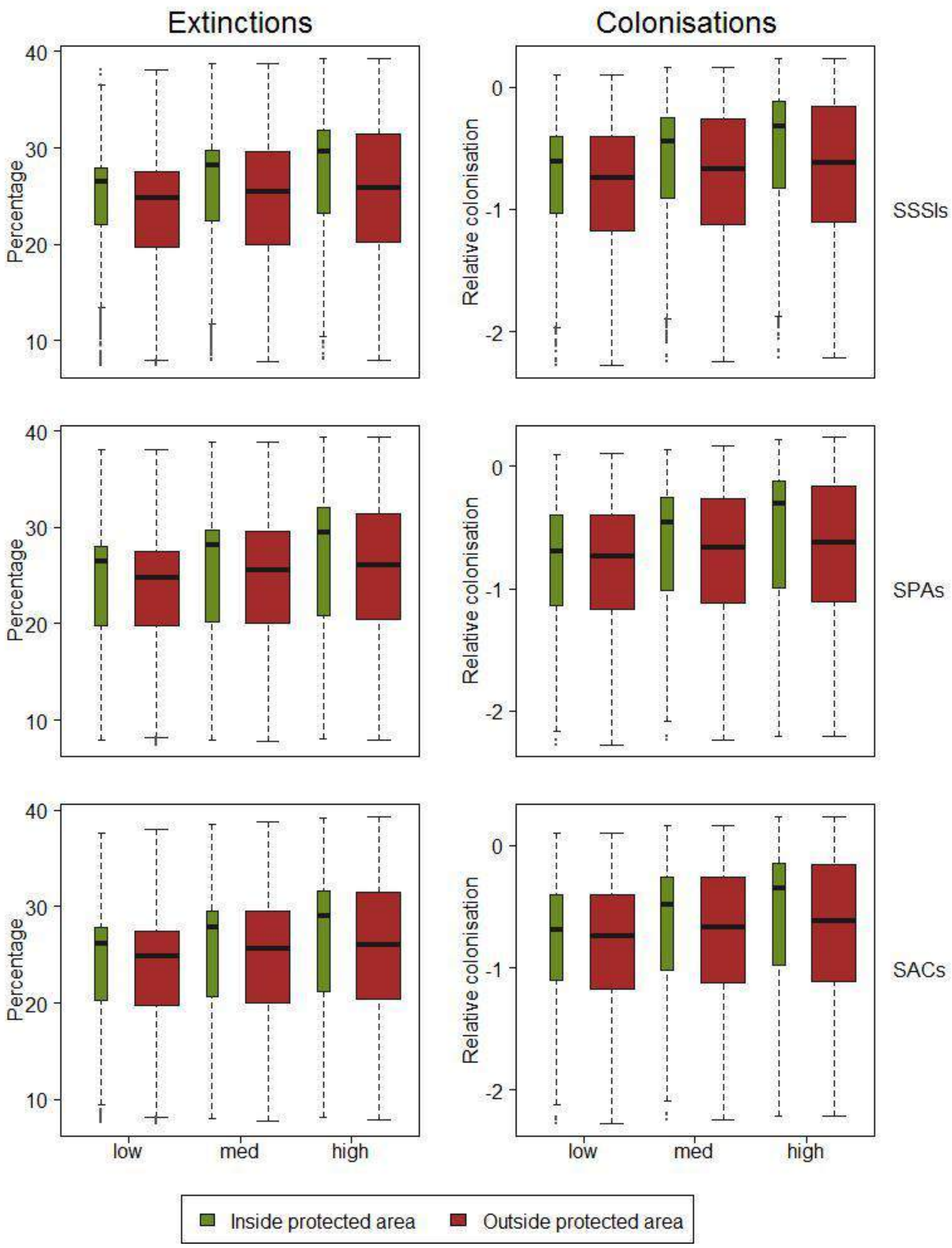


Figure 4-14 Percentage of extinctions, and relative colonisations under *2*°C *, 4*°C *and 6*°C climate scenarios. Box widths are relative to number of species in each group.

(Inside and outside the existing protected area network. For colonisations, more negative values equate to a lower relative colonisation.)

Priority areas for new conservation action

4.1.42 Zonation analysis, seeking to identify sites that could either enlarge existing protected areas, or create new ones to maximise the representation of all UK wildlife within the protected area network, suggested that the current proportion of species adequately protected within the SSSI (Figure 4-15), SPA (Figure 4-16) and SAC (Figure 4-17) protected area networks could be improved by selecting new areas for further protection. For the SSSI and SAC protected area networks, a 6°C scenario suggested an increase in protected areas in Northern Scotland, the western isles and Northern Ireland would improve future protection for all species combined.

4.1.43 Whereas across the SPA (Figure 4-16) network, a 6°C scenario suggested that changes to the network in broadly similar areas to current priorities would also improve protection for bird species in the future Judicious selection of new areas to protect could result in improved protection for UK wildlife both now and in the future.

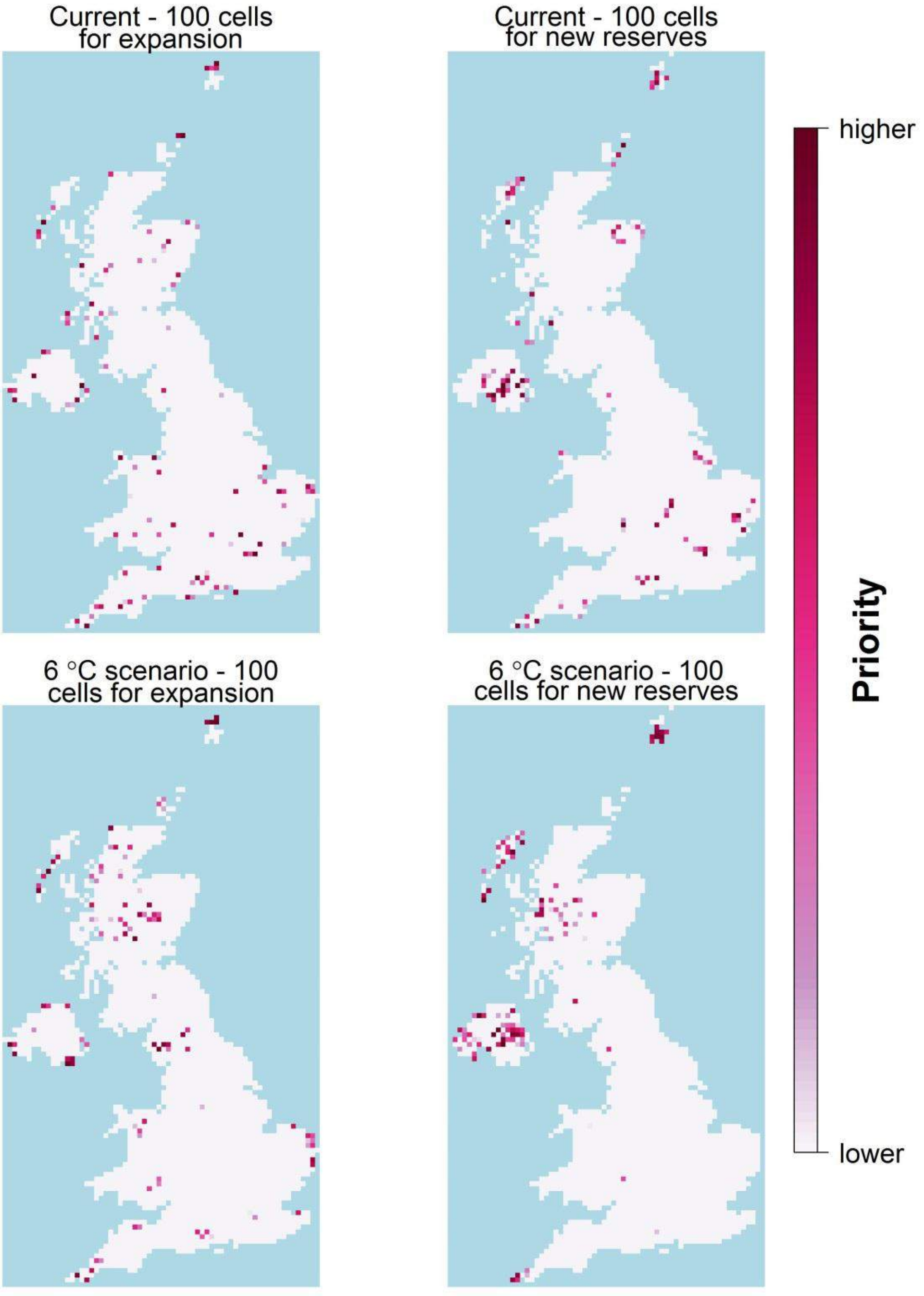


Figure 4-15 Priority areas for expansion of existing SSSIs and for new SSSIs for current climate and for the 6°C climate scenario

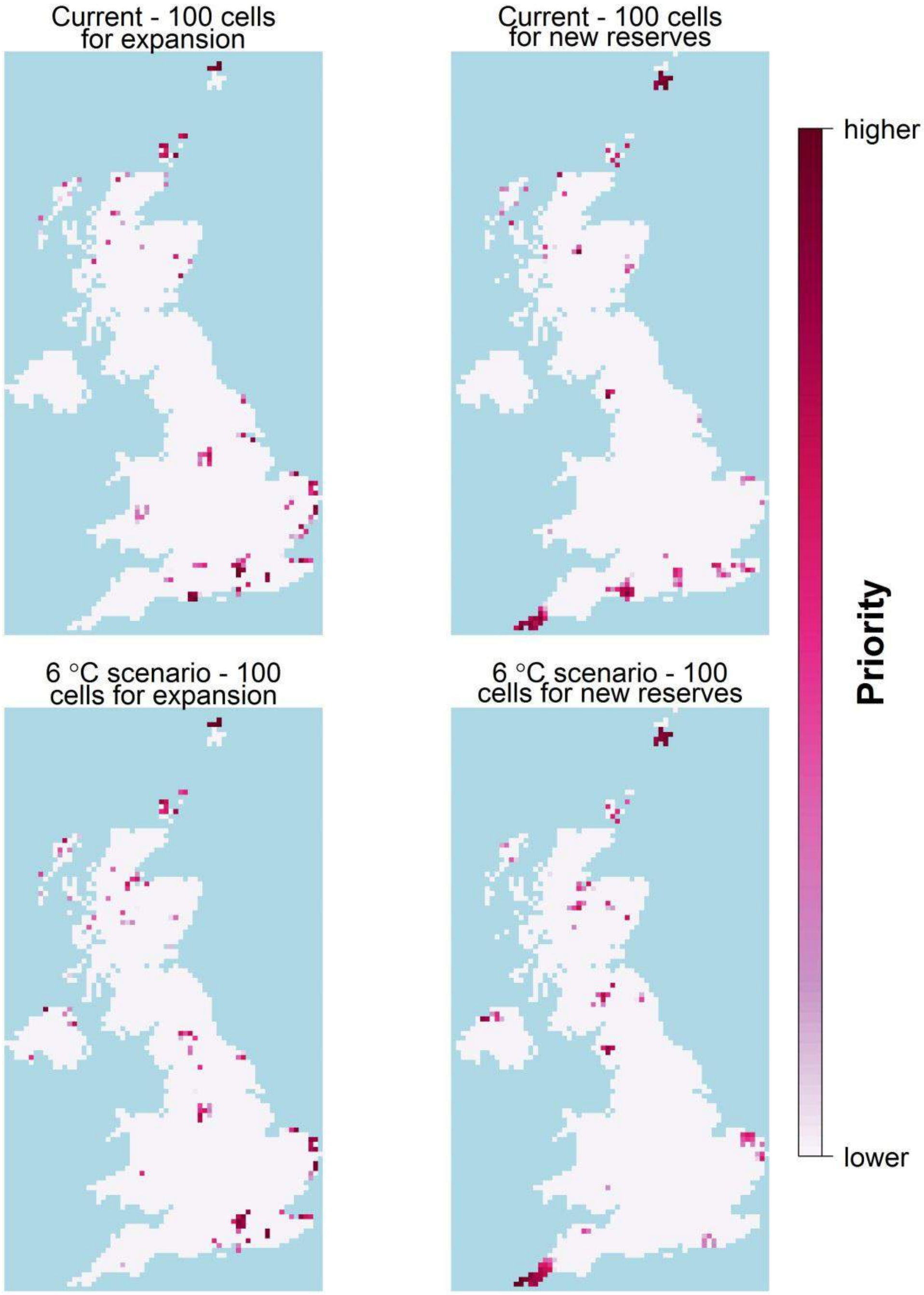


Figure 4-16 Priority areas for expansion of existing SPAs and for new SPAs for current climate and for the 6°C climate scenario

(These SPA results only include use of bird species in Zonation analysis.)

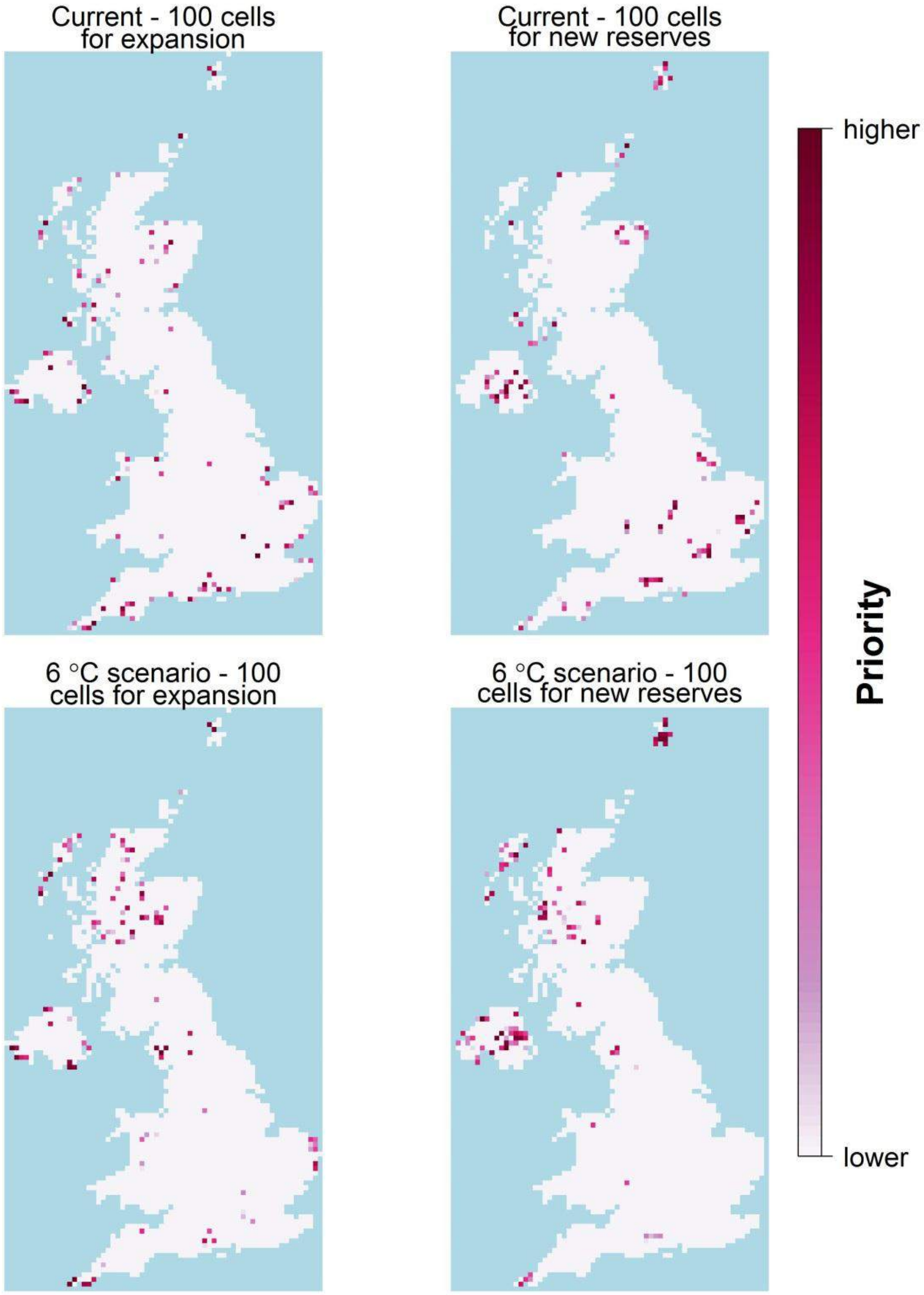


Figure 4-17 Priority areas for expansion of existing SACs and for new SACs for current climate and for the 6°C climate scenario

4.1.44 For most definitions of 'adequately protected' (i.e. the number of 10km cells where species have climate space remaining), steep gains in the proportion of species meeting the criteria can be realised by relatively small increases in the extent of protected area in the UK if the sites for expansion and newly protected sites follow the suggestions identified above (Figure 4-18). An example of the gains in proportion of species protected under a definition of adequate protection for UK species being 50 10km grid cells is shown in Table 4-2. For example, increasing coverage of SSSIs in the future under a 2 change from 10% to 17% could increase UK species °C protection by 6%. We recommend that the results of this Zonation analysis be viewed as a demonstration of concept, showing that developing a full spatial conservation plan for the UK has the potential to make significant contributions to wildlife protection both currently and under likely future climate conditions.

Table 4-2: Proportion of species protection under protected area networks when the definition of adequately protected is 50 10km grid cells. The future scenarios are under a 2°C climate change. Similarities among SSSIs and SACs are due to high overlap of these protected area networks.

		Percentage of area protected						
Climate scenario	**Protected area network**	**5%**	**10%**	**17%**	**25%**	**50%**	**75%**	**100%**
Current	**None**	0.56	0.73	0.80	0.83	0.88	0.90	0.91
2°C change	**None**	0.60	0.68	0.74	0.77	0.84	0.86	0.87
Current	**SSSI**	0.58	0.73	0.80	0.82	0.88	0.90	0.91
2°C change	**SSSI**	0.60	0.67	0.73	0.77	0.83	0.86	0.87
Current	**SPA**	0.44	0.60	0.66	0.69	0.75	0.75	0.75
2°C change	**SPA**	0.43	0.58	0.63	0.69	0.74	0.76	0.77
Current	**SAC**	0.58	0.73	0.80	0.83	0.88	0.90	0.91
2°C change	**SAC**	0.60	0.68	0.73	0.78	0.84	0.86	0.87

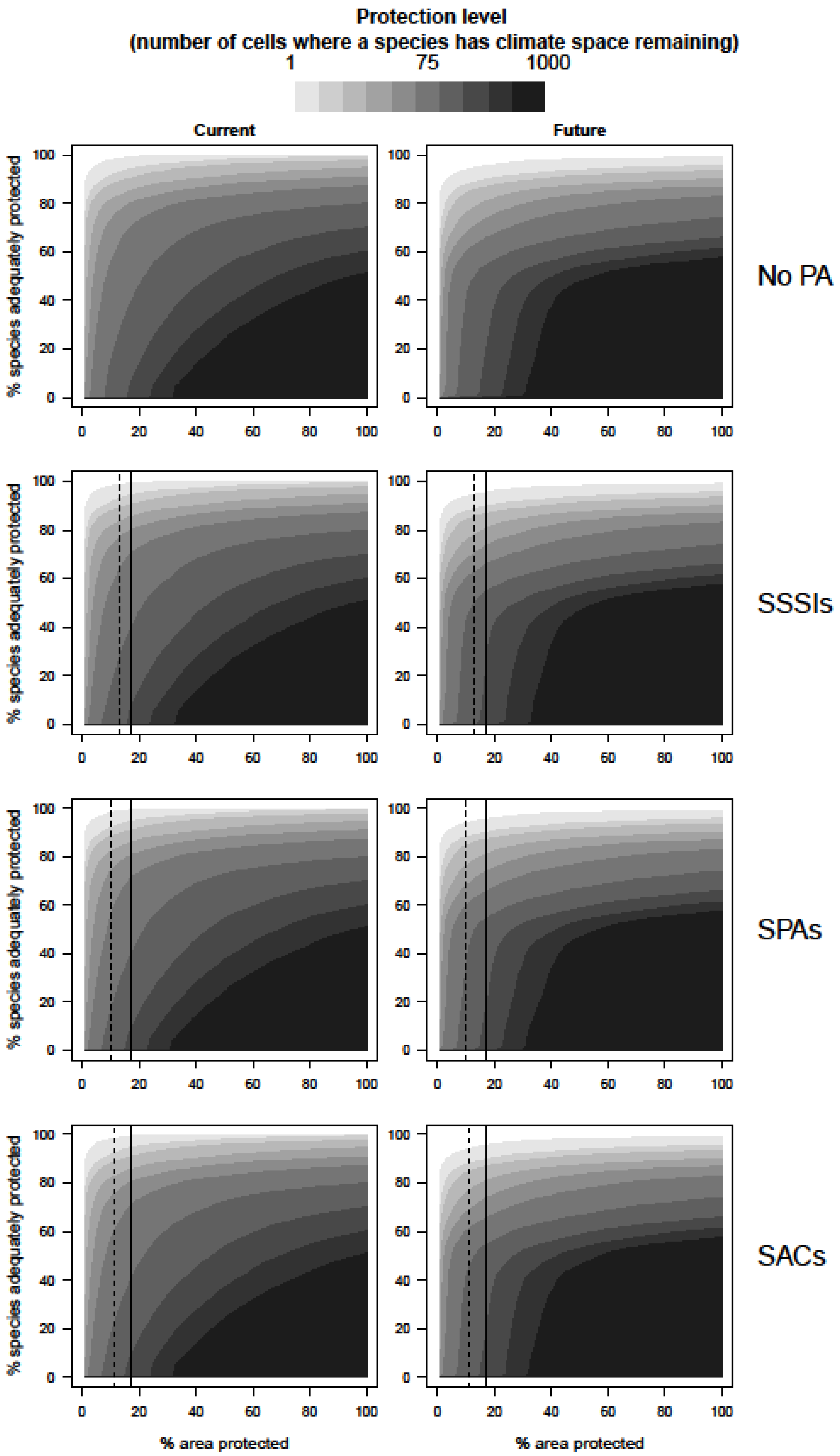


Figure 4-18 Variation in percentage of species protected under different definitions of adequately protected.

(i.e. the number of cells where a species has climate space remaining within the protected area network that is used to define adequately protected) under the current and low climate scenario. Dashed line indicates the percentage of cells with more than 5% coverage under current protected area network and the solid line indicates the 17% Aichi target of protected area coverage.)

Climate risk

Comparisons across climate scenarios

4.1.45 Future projections of climate change were based on UKCP09 projections for 2070-2099 for B1, A1B and A1F1 models, equivalent to approximately 2°C, 4°C and 6°C scenarios of global warming. This time period was used because it represents extreme values for projections; between the current and this extreme we would expect intermediate values for projections. This comparison enables an assessment of likely impacts of an increasing magnitude of climate change upon species in the UK to be made. These results show that species impacted by a 2˚ global warming scenario are even more impacted by the 4˚ scenario (as measured by changes in the number of 10 x 10km squares each species was projected to occupy), with almost perfect correlation between the results (Figure 4-19). However, the difference between the two scenarios appears small, relative to the changes projected to occur between the current climate and both future projections. This appears to be because the two global scenarios produce smaller differences locally within the UK, with most changes having already occurred under a global 2°C change scenario. Species that show large range expansions are likely to reach the far north of Scotland under the low scenario, so cannot show further change under higher scenarios. The same pattern occurred when comparing low and high climate scenarios; species impacted by a 2˚ global warming scenario are even more impacted by the 6˚ scenario (Figure 4-20)

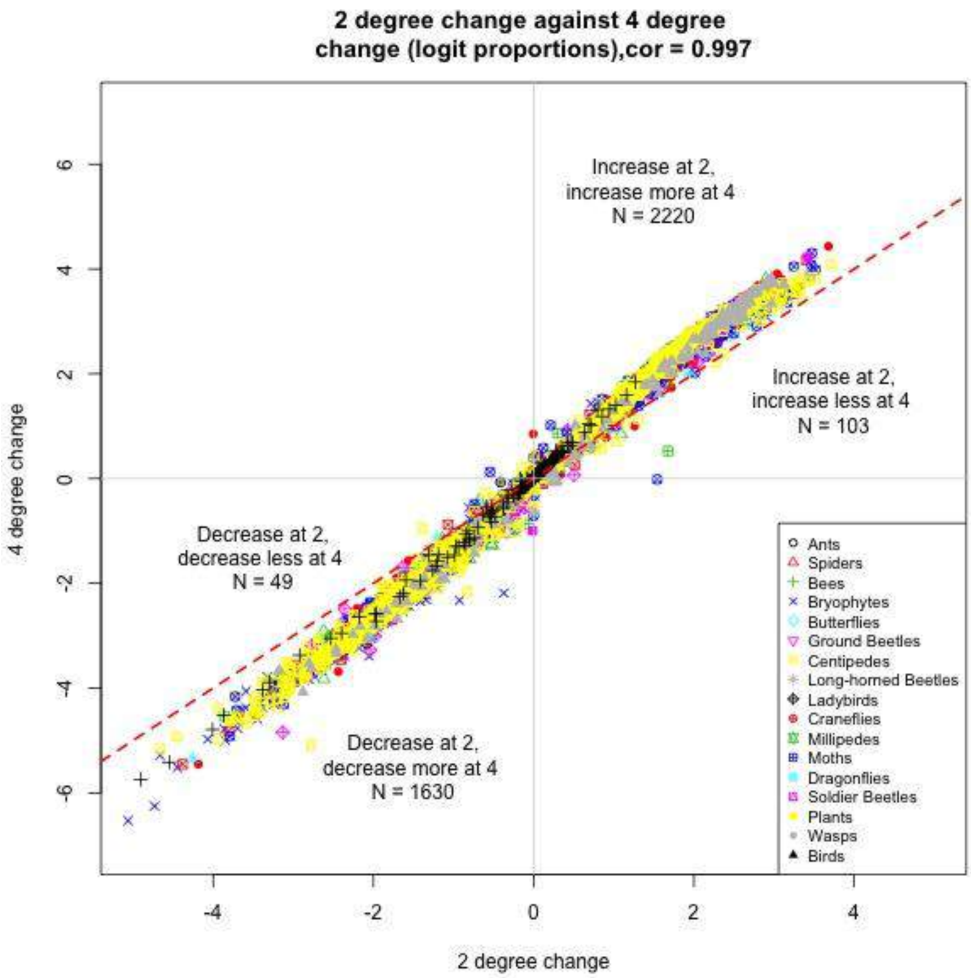


Figure 4-19 Projected distribution changes under 2 and 4°C global warming scenarios

(Raw changes in probabilities have been converted to changes proportional to the available space for change: increases are expressed as a logit proportion of currently unoccupied space, decreases as a proportion of currently occupied space.)

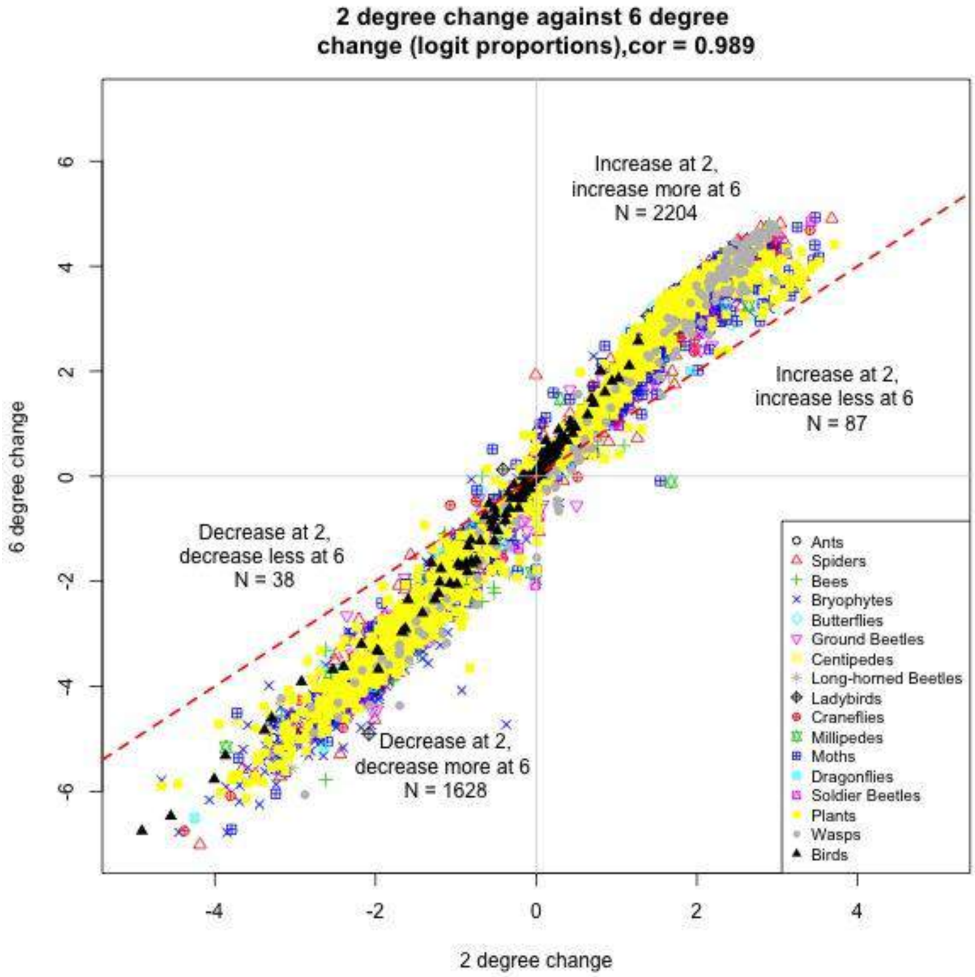


Figure 4-20 Projected distribution changes under 2 and 6°C global warming scenarios

(Raw changes in probabilities have been converted to changes proportional to the available space for change: increases are expressed as a logit proportion of currently unoccupied space, decreases as a proportion of currently occupied space.)

Relative importance of temperature and rainfall

4.1.46 In Figure 4-21 we show the relative importance of the four climatic variables determining distribution across all taxonomic groups. Overall, the importance of climate variables is similar among the groups. Wasp and bird species distributions are most sensitive to MTCO (mean temperature) relative to other groups. Birds are also most sensitive to cvTemp (seasonality). Sensitivity to GDD (summer warmth) was similar across the groups, while ants are the group least sensitive to soilWater (soil moisture). Overall, there is also variation in the relative sensitivity of species groups to each climate variable. For example, ants are more sensitive to summer warmth in comparison to other species groups, but least sensitive to seasonality and soil moisture levels.

4.1.47 It should be noted that these associations reflect long-term climatic associations, not short term weather impacts: species that are apparently sensitive to long term fluctuations in seasonality are likely to be responding to indirect impacts of seasonality, and equally those showing sensitivity to temperature may, for example, be distributed in cold areas, but actually breed and survive well in warm years.

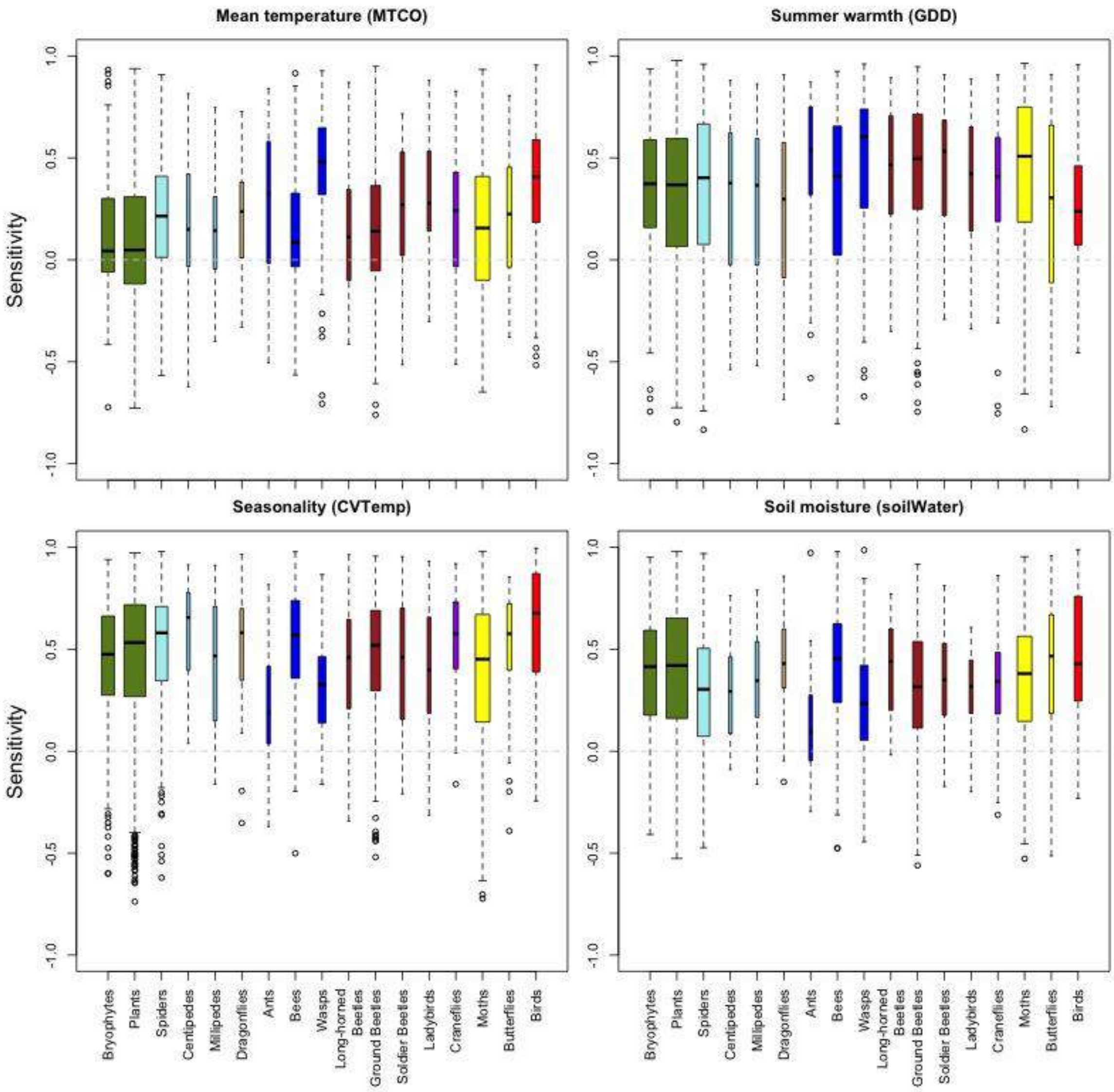


Figure 4-21 Relative importance of the four climate variables under the low climate scenario

Climate specific results

Distribution of climate variables

4.1.48 The most important changes of variation in climate variables occurs between current climate and the low scenario, while there is lower variation between the low, medium and high climate scenarios (Figure 4-22).

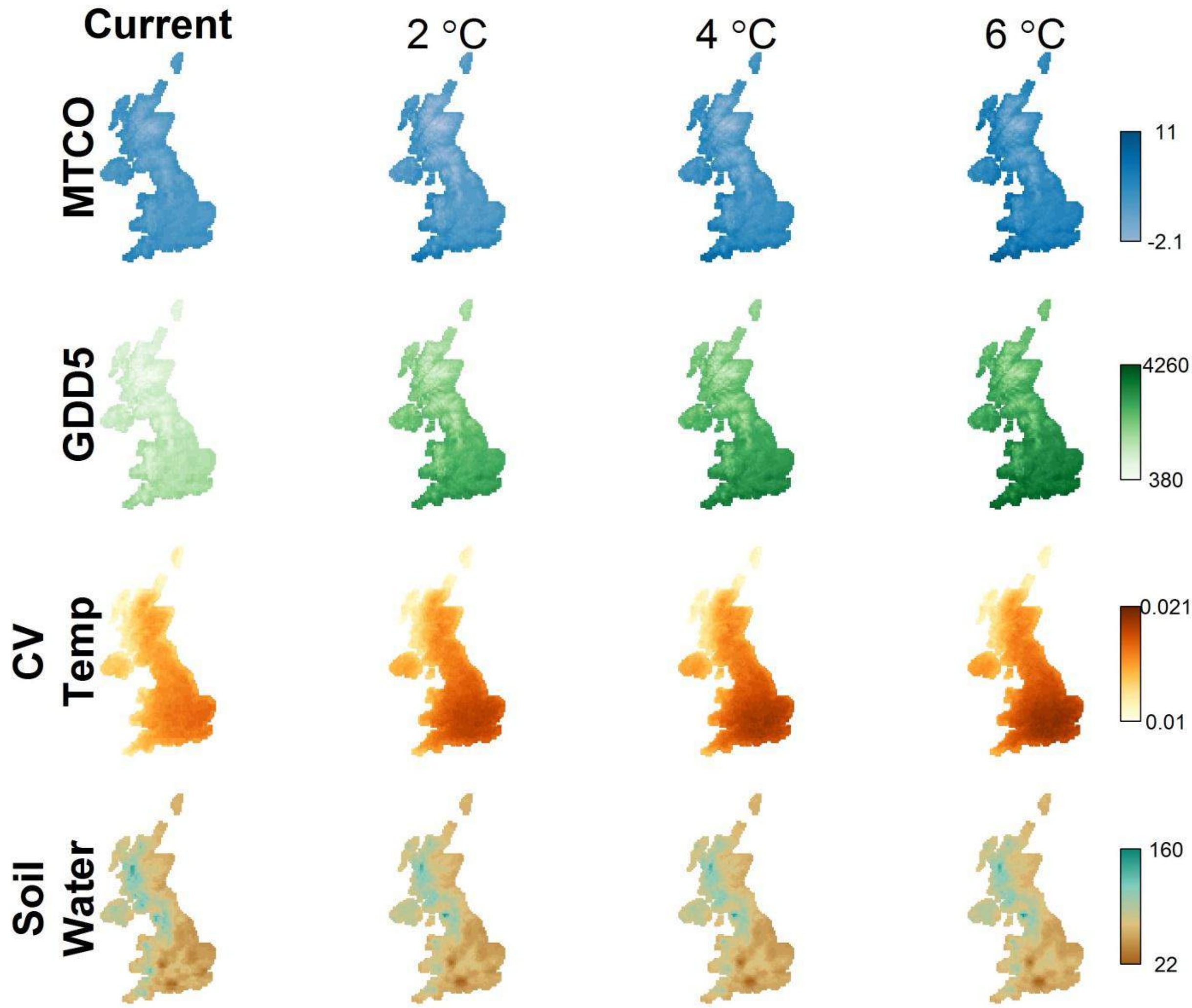


Figure 4-22 Variation of climate variables between current climate and future climate scenarios

(MTCO, mean temperature of the coldest month (OC); GDD5, growing degree days (index of temperature above 5Oc); cvTemp, temperature variation (mean monthly temperature/standard deviation); SoilWater, and index of water balance.)

Maps for climate only results

4.1.49 One of the advantages of the method we used is that as well as the above results that assume all other processes influencing species distribution (i.e. non-climatic processes) remain the same, we can explore the results if we assume all additional constraints are removed. For example, in our models, the distribution of the Common Buzzard, *Buteo buteo* is relatively poorly explained by climate and other processes are very important. This is probably a consequence of persecution across most of England at the time the data were collected: recently persecution of this species has eased and the species has once again spread across much of England. Our models correctly identified this species as sensitive not to climate, but other factors and, once action had been taken in the other areas, the species should spread to ultimately fill its climatic niche. Assuming equally effective action could be taken for all other species (clearly unlikely in practice, but useful to show where other factors are most important), we can model the patterns of colonisation and extinction for species again.

4.1.50 These results for percentage of local extinctions (Figure 4-23) and numbers of local colonisations (Figure 4-24) show broadly similar patterns to the full analysis above. Areas highlighted as most different for extinctions include Northern Ireland and upland areas in South West England, the Lake District and north west highlands of Scotland. For colonisations, the greatest differences are seen in the Scottish islands which have a greater number of colonisations in comparison to the results taking into account non-climate processes. These areas could be heavily impacted by agricultural practices such as intensive grazing, which is likely to have a significant impact on biodiversity.

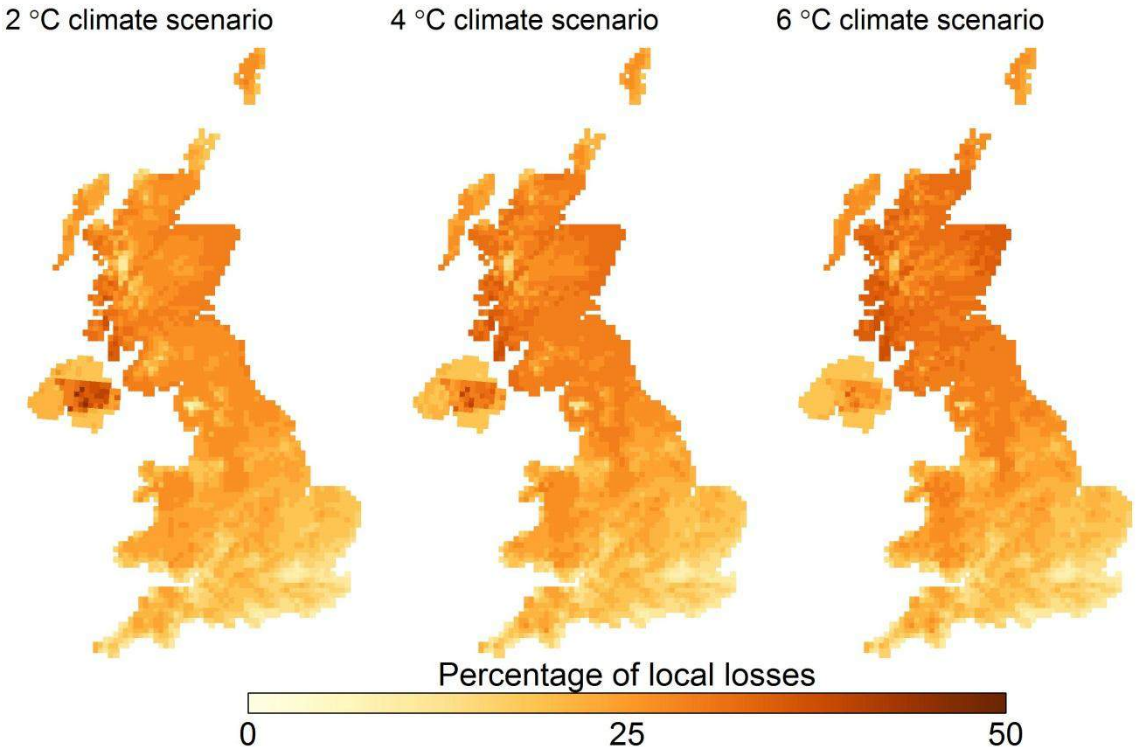


Figure 4-23 Areas likely to see local extinctions

(i.e. local loss of species per 10km grid cell for 2°C , 4°C and 6°C climate scenarios using climate only variables.)

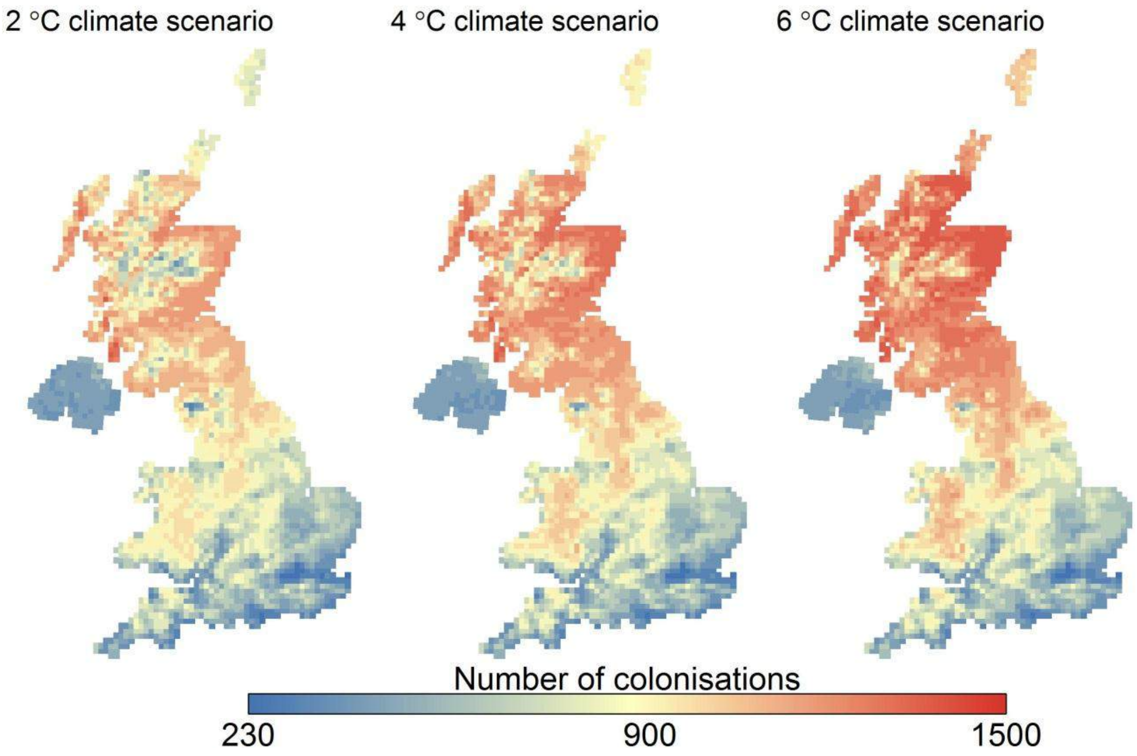


Figure 4-24 Areas likely to see colonisations from existing UK species

(per 10km grid cell for 2°C , 4°C and 6°C climate scenarios using climate only variables.)

Summary of findings

4.1.51 The key findings from the wildlife modelling were:

- High species-specific sensitivity to climate change, but systematic differences between taxonomic groups.
- Relatively little difference in the proportions of species in different risk categories associated with 2, 4 or 6˚C global warming scenarios projected to 2070-2099 in comparison to changes against the 1960-90 baseline: most of the changes happen in the UK with relatively little climate change.
- Bryophytes are one the groups most likely to be negatively impacted, and also of high international conservation importance.
- Greatest change in northern and upland areas (but no data on new colonists to the south).
- Protected areas are likely to see relatively more species turnover in comparison to surrounding land.
- Results assessing the impact of climatic variables only generates broadly similar patterns to the full analysis.
- Additional protected areas, particularly in the north west of the UK, could potentially improve species protection in the future.

4.1.52 The key findings for the constituent parts of the UK were:

- England – A contrast between northern and southern areas of the country, with northern areas expected to see greater colonisations and extinctions. However, the analysis does not account for colonists to the south. A high number of additional SPAs could improve protections for bird species in the future, particularly on the coast of East Anglia and the south west.
- Northern Ireland – In comparison to the other countries there is likely to be the lowest numbers of colonization and extinctions. For all three protected areas networks, a high number of additional protected areas, particularly on in coastal areas, are likely to improve species protection for UK species.
- Scotland – Mainland areas are likely to see the highest numbers of colonisations and extinctions (i.e. greatest turnover) in comparison to the rest of the UK, particularly on the west coast. In contrast, the western and northern isles have some of the lowest expected species turnover. Zonation analysis for the whole of the UK suggests that the majority of new protected areas in the future should be located in Scotland.
- Wales – Upland areas are likely to see the greatest levels of colonisations and extinxtions, with coastal areas having a relatively lower species turnover. Prioritising areas for new and expnansion of protected area networks across the UK resulted in very few additional priority areas located within Wales.

Caveats

4.1.53 The following key caveats apply:

- These analyses are modelling the potential changes in climate space available to UK biodiversity, not the actual possible change in distributions of UK species.
- These analyses do not take into account indirect impacts of climate change on biodiversity through other mechanisms such as land use policies.
- Other factors that influence distribution of species such as habitat type and connectivity, species interactions and behavioral and phonological changes are not reflected in the analyses.

- The results do not include species that are not yet present in the UK, and therefore will underestimate total colonisation for the species from more southerly regions of Europe that are shifting polewards due to climate change. In addition, information in soil biodiversity is not included in the analyses.

- Uncertainty of results increase at higher climate scenarios, and in some areas is already quite high.

- Species groups without European distribution data (i.e. not plants or birds) are likely to be subject to greater errors and uncertainty in model outputs.

- These analyses do not take into account presence of refugia; localised environments that provide protection from broader-scale climatic changes.

Valuation

Overview

4.1.54 The original aim of this section was to develop estimates of the monetary value of changes in the density and distribution of wildlife under a range of future climate scenarios and based on the modelled outcomes (Section 0). However, because of the conceptual difficulties in valuing wildlife (see discussion below), and particularly the non-use values of wildlife, it was agreed that no such attempt should be made.

4.1.55 The sections that follow are therefore limited to a discussion of some of the conceptual and methodological difficulties with valuing wildlife, the approaches that could be adopted, the limitations of these and two case studies that serve to highlight the significant contribution (in terms of use value) that two specific economic sets of species (pollinators and wild game) presently make to the economy. It is, however, important to note that the value of changes in well-being (particularly relating to use values) as a result of the effects of climate change on wildlife is context specific and will depend on *inter alia* the current stock (abundance and richness) of species, people's economic preferences for these and the extent of loss or gain of different species as a result of climate change.

Valuing wildlife

4.1.56 Wildlife has both use and non-use value. Use values are defined as the value derived from the actual use of a good or service and include the role of wild species in the direct delivery of ecosystem services such as pollination (see Box 4.1), fertilisation and pest reduction which are inputs to crop production; maintaining genetic diversity; providing opportunities for leisure, recreation and tourism (e.g. bird watching and shooting – see Box 4.2); and in contributing to educational and ecological knowledge.

4.1.57 Quantifying the use value of wild species is relatively straightforward, and there are a number of studies that have looked at the benefits that wild species provide in terms of tourism, recreation, hunting, and pollination benefits etc. Box 4.1 and Box 4.2 provide some examples of the direct use values of wildlife in relation to pollination and recreation, respectively.

Box 4.2 The economic value of wild pollinators

It is well known that pollinating insects play a crucial role in providing pollination services to agricultural crops (Klein *et al*. 2007; Zhang *et al*. 2007; Defra, 2013). Pollination is either abiotic, primarily by wind; or biotic, primarily by bees and other insects. Several wild pollinators are listed in the UK Biodiversity Action Plan (BAP): these include 20 from a total of about 250 species of bee, 24 from a total of 56 species of butterflies and 7 of 250 species of hoverflies and many other insects. Since 1980, wild bee diversity has declined in most landscapes, with habitat and diet specialist species suffering greater losses than more generalist species

(Biesmeijer *et al*. 2006); hoverflies showed both increases and decreases in diversity for the same time period, but again specialists fared poorly. Butterflies, though rarely pollinators in the UK, have also undergone major range and population shifts (Asher et al. 2001). While there is evidence of pollinator losses in the UK (Biesmeijer *et al*. 2006), there is very limited understanding of the consequences of these losses for pollination services. Consequently, a £10 million research programme is currently underway to address these knowledge gaps (see www.bbsrc.ac.uk/pollinators).

Research conducted for the UK NEA estimates that 20% of the UK cropped area in 2007 comprised pollinator-dependent crops and notes that a high proportion of wild, flowering plants depend on insect pollination for reproduction. The relationship between pollinators and service delivery is not well understood, but the few available studies indicate that pollinator diversity is linked to greater crop yield, resilience and stability (Hoehn et al. 2008; Winfree and Kremen 2009).

Several studies have demonstrated that locally deficient pollinator communities can lead to reduced crop yield and/ or quality (Ricketts et al. 2008). Decreases in pollination services would, therefore, result in short-term economic losses for famers, at least until alternative windand self-pollinating crops replaced insect-pollinated crops, or supplemental services could be brought in through managed pollinators. It would also force a change in food choice and security in that UK consumption would either have to shift away from pollinator-dependent products, or a greater reliance would have to be placed on imported pollinator-dependent foods.

Climate change is predicted to result in declines in European bee species richness (Dormann et al. 2008); though expected shifts in the UK are not projected, disruption of plant-pollinator networks may be expected (Memmott et al. 2007). Bumblebee declines in the UK have been related to climatic niche shifts (Williams et al. 2007). Continued declines, or total loss of one or more pollinator functional groups, would be expected to have direct and short-term consequences for agricultural producers and consumers and wide-ranging and longer-term impacts on wild plant communities and wider ecosystem functions. The value of total (biotic) pollinator loss for UK agriculture was estimated to be around £430 million in 2007 (England: £364 million, Northern Ireland: £19 million, Scotland: £47 million, Wales: unknown), which is approximately 8% of the total value of the market (Gallai *et al*, 2009 using data from Defra 2008, BHS 2008).

However, this estimate fails to take into account the contribution of pollinators to forage crops, such as clover, which support livestock; small-scale agriculture, such as allotments and gardens; ornamental flower production; and seed production for agricultural crop planting. Furthermore, as the majority of wild flowering plants depend upon insect pollination, decreases in pollinators will also result in a reduced seed/fruit set and may ultimately lead to the local extinction of plant species (Ashman et al. 2004; Aguilar et al. 2006). Loss of flowering plants will reduce the availability of resources for pollinators, which, in turn, will reduce insect pollination services for plants in a positive feedback loop (Bascompte et al. 2006). The value of pollinators and pollination services to wildflowers and for recreational and other cultural services is unknown, but is expected to be significant. Several studies indicate that diverse, visible assemblages of wildflowers make important contributions to the aesthetic qualities of whole landscapes and roadside verges within the UK (Willis and Garrod 1993; Akbar 2003; Natural England 2009). So, while the figure of £430 million per year would be a conservative estimate of total loss value, it is an unrealistic scenario as complete pollinator collapse in the UK is improbable. Indeed, the modelled outcomes in Section 0 suggest that although there are likely to be some local extinctions, the spatial range of bees and other pollinators is likely to expand across each of the climate scenarios investigated.

For the purposes of policy, the value of lost production in certain areas as a result of local extinctions may also be contrasted with the costs of supporting the provision of natural and semi-natural habitats such as woodlands and semi-natural grasslands, including through agri-environment schemes, which provide a wide range of flower communities and could therefore support bees and insects and which have been shown to provide a spill-over of pollinators into farmland and can increase pollination services (Tinch, 2010; Kremen et al. 2007; Ricketts et al. 2008).

In order to estimate the change in agricultural production as a result of a change in pollinating insects, the following information is required:

- the current contribution of pollinators to agricultural output (by crop type) across the UK;
- the current distribution of pollinator-dependent crop types across the UK;
- the change in abundance and/or distribution of wild (insect) pollinators as a result of climate change; and
- the relationship between pollinator abundance and crop productivity.

Box 4.3 Some examples of the contribution of wildlife to the value of tourism and recreation

The direct appreciation of wildlife can generate substantial benefits, as evidenced by the widespread participation in activities such as bird watching, angling and wildlife photography. For example, the UKNEA reports that licensed anglers fished a total of 30 million days during 2005: about 26 million for coarse fishing and 4 million for game (salmon and trout) fishing (Bateman et al. 2011). It has also been estimated that there were at least 2.9 billion leisure visits to natural environment settings between 2009 and 2010 (Adaptation Sub-Committee 2013).

The benefits that people derive from these activities may be valued through observed behaviour (e.g. applying the travel cost method to estimate expenditure on wildlife-watching trips as a proxy for their value, or estimating values through licences or membership fees). Novel research undertaken for the NEA combined a geographic information system (GIS) with data obtained from the Monitor of the Engagement with the Natural Environment (MENE) survey to model how the distribution of natural environment and urban resources interact with population distribution in determining recreational visit flows (Sen et al, 2011). The visit flows were then valued on the basis of a meta-analysis of over 200 previous estimates of the value of a recreational visit, examining the influence of the environmental characteristics of visited sites and the differences in the methods used to generate those value estimates. While this provides a useful spatially disaggregated estimate of the recreational value of different habitat types, it does not say anything about the relative contribution of particular wildlife species to the total value. As such, any values will provide only a weak proxy for the value of wildlife.

4.1.58 The non-use values people may hold for wildlife include the existence value of species (i.e. the knowledge that wild species are being conserved even if the individuals expressing that value do not observe the species concerned) and bequest values associated with knowing future generations may benefit from the continued existence of these species. Fuller et al. (2007) and Weinstein et al. (2009) have also demonstrated the health and well-being benefits from engagement with biodiversity where benefits may be positively correlated with species richness or other dimensions of biodiversity.

4.1.59 While there is substantial anecdotal evidence of non-use (existence and bequest) values associated with maintaining biodiversity, obtaining reliable estimates of associated values is somewhat problematic. This is because, unlike use values, it is not possible to observe behaviour regarding non-use values (e.g. actual payments), neither are they reflected in productivity. While it is arguable that a lower boundary estimate of values might be provided by the payments provided by policies designed to promote biodiversity (e.g. the profits forgone by farmers when they agree to take on biodiversity-focused agri-environment schemes or the costs of implementing the UK Biodiversity Action Plan), there is a risk that this leads to potential circularity of the valuation process (Bateman et al. 2011).

4.1.60 An alternative approach is to elicit individual preferences for the non-use value of biodiversity through stated preference studies.[15] UK-level estimates of the non-use (existence) value of terrestrial biodiversity elicited through stated preference studies range from £540 million to £1,366 million per annum (Boatman et al. 2010; Christie et al. 2010). There is nevertheless considerable debate regarding the robustness of monetary estimates of the non-use (existence) value of biodiversity derived through stated preference methods. While conforming to conventional economic principles for non-market environmental goods for which individuals hold well-formed economic preferences, commentators disagree over whether preferences for the non-use (existence) value of biodiversity conform to these requirements (see Bateman et al. 2008; Morris and Camino 2010; Bateman et al. 2011). Moreover, such methods are resource intensive, requiring significant periods of time and money to implement properly, and are therefore seldom feasible for the purposes of supporting policy decisions.

4.1.61 There are, however, increasingly sophisticated techniques to support the transfer of values from an original source to new policy situations. This value transfer can either make use of a single estimate from a prior source valuation study that values the same good in a context that approximates that of the policy application and is then transferred (with relevant adjustments) to the latter or, where more substantial evidence is available, using the results from a meta-analysis to relate values to the characteristics of previous studies and the goods and services valued. Nevertheless, the application of value transfer approaches requires suitable evidence from which value estimates can be drawn. Ideally, this evidence should approximate the situation to which the values are to be transferred in terms of the type of good being valued, the scarcity of that good and the nature of the change (in availability or quality) being valued.

4.1.62 In the absence of such evidence, Bateman et al. 2014 in the UKNEA Follow-On (UKNEA-FO), consider an alternative approach which focuses on the costs of ensuring specified levels of provision (i.e. ensuring no net loss). This approach examines the opportunity cost (e.g. the foregone income from agricultural production) of implementing measures to prevent land use change in order to protect critical habitats. However, as Bateman et al. point out, such an approach is not a valuation of the wild species concerned. Rather, it provides an indication of the nature and significance of trade-offs involved and can be used to initiate payments for ecosystem services (PES) schemes whereby the ‘providers’ of the service (the habitat in which wildlife resides) are rewarded by the beneficiaries of that service for its continued provision.

4.1.63 Mourato et al. (2010) have also examined an alternative way of measuring the non-use value of biodiversity by examining actual payments for non-use-related wildlife conservation in the form of legacies to environmental charities. Legacies can be argued to represent a pure non-use value: individuals leaving a charitable bequest to an environmental organisation in a will, for the purposes of supporting their conservation activities, will not experience the benefits of this work. Overall, the total legacy income earned by environmental charities in the UK in 2008/09 was £97 million (Mourato et al. 2010). However, as noted by Bateman et al. (2010) using legacy payments as a proxy for non-use values captures only one element of environmental non-use values, i.e. those that are reflected in the marketplace at the time of death. Moreover, very little is known about charitable bequests in the UK. Data on charitable bequests, estates and demographic characteristics of donors is not easily accessible, particularly for analysis over time. Equally, comprehensive data on charitable giving over time, from the perspective of the recipient organisations, and covering a wide range of organisations, is not freely available.

[15] These make use of surveys to ask individuals to make choices between different levels of environmental goods at different prices to reveal their willingness to pay for those goods.

4.1.64 While these non-use values are likely to be substantial, the modelling outcomes presented in Section 0 suggest that although some species are highly sensitive to climate change (particularly bryophytes and ants) and there will be some local losses, in most cases climate change will result in a redistribution of species across the UK. The effects of a redistribution of species on non-use values cannot easily be determined but are likely to be considerably smaller than the effects of a total (UK-wide) loss.

Conclusions

4.1.65 Wildlife is highly sensitive to climate change and its ability to adapt to changing weather patterns is, at least in part, affected by the way land, in particular semi natural habitat, is managed (Adaptation Sub-Committee 2013).

4.1.66 Given the difficulties in deriving robust values for the non-use values of wildlife and the fact that focusing on economic species (or some sub-set thereof) would only give a partial picture of the change, no attempt has been made to value the impacts of climate change on wildlife in light of the wildlife modelling.

4.1.67 Valuing the impacts of climate change on wildlife requires:

- access to established relationships between species abundance, species distribution and productivity (see Box 4.1);
- knowledge of threshold effects (i.e. the point at which the population of a wild species is no longer sustainable and its use value ceases altogether);
- availability of current and future species abundance, distribution and productivity, how this is likely to change in response to both climate signals and other climate-induced changes in ecosystem functioning and service provision; and
- robust marginal value estimates for non-use values of individual species that can be applied to the scenarios being investigated.

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Appendix: A

The Environment Agency's 'Reasons for not achieving good' dataset for England and Wales.

This section provides brief descriptions of the sectors, pressures and sources of the SWMIs in the 'Reasons for not achieving good' dataset, followed by the numbers behind Figures 8, 9, and 10 in the main report.

A Significant Water Management Issue (SWMI) refers to an issue that will need to be addressed to achieve environmental objectives under the Water Framework Directive.

Sectors

- **Water industry:** Examples include the transfer of water from one river system to other for public water supply (flow, surface water abstraction), sewage discharges (these are generally permitted).
- **Agriculture and rural land management**: Examples include livestock farming leading to bank poaching or sediment load modification (physical modification), or impoundments for water abstraction for agricultural uses.
- **Urban and transport**: Examples include runoff from hard surfaces in residential areas (diffuse, drainage – housing), or physical modification for development in urban settings.
- **Sector not relevant**: This can be ascribed when the failure is due to natural means (e.g. levels of copper, barriers to fish), or the presence of invasive non-natives, Still under investigation: Sector still to be identified. These are often physical modifications.
- **Mining and quarrying**: Often diffuse or point source pollution from abandoned mines.
- **Industry, manufacturing or other business**: This covers a variety of impacts, including point source discharges from manufacturing, physical modifications creating barriers to fish movement, and landfill leaching affecting ammonia and dissolved oxygen concentrations.
- **Central Government**: Mostly land drainage or flood protection leading to physical modifications, or barriers to fish.
- **Domestic/general public**: Point source sewage discharges, or unsewered. Some barriers to fish movement.
- **Other**: This is not a sectorial category in the 'Reasons for not achieving good' dataset; rather is a grouping category added by the Project C team. Sectors included are: navigation; angling and conservation; local government; recreation; waste treatment, transfer, storage, disposal. Combined they represent just less than 2% of the provenance of failures.

Appendix Table 1 The sectorial provenance (% share) of SWMIs in the Environment Agency's 'Reasons for not achieving good' dataset.

Sector	Share (%)
Water industry	30.5
Agriculture and rural land management	28.4
Urban and transport	15.8
Sector not relevant	6.0
Still under investigation	5.4
Mining and quarrying	4.2
Industry, manufacturing and other business	3.7
Central Government	2.9
Domestic/general public	1.2

Sector	Share (%)
Navigation	0.9
Angling and Conservation	0.4
Local Government	0.4
Recreation	0.2
Waste treatment, transfer, storage and disposal	< 0.1

Pressures

- **DOC (Dissolved Oxygen Concentration**): Although high DOC levels can be harmful in some contexts, the WFD defines status based on low concentrations being 'bad' (and 'poor' or 'moderate') and high concentrations being 'good' or 'high'. This is based on the conditions required for salmonid fish. Many DOC-related failures are due to point source sewage discharges or diffuse source agriculture.

- **Ammonia**: High levels of ammonia can cause WFD failures due to the potential for harm to fish (or invertebrates). Because of this, ammonia is listed as a specific pollutant under Annex VIII of the WFD, but it is cited separately to other specific pollutants in the 'Reasons for not achieving good' dataset.

- **BOD (Biochemical Oxygen Demand)**: The amount of oxygen required for the decomposition of organic matter present in the water. BOD is considered to be a good proxy for organic pollution, and most failures under this criterion arose from point source sewage discharges or diffuse agricultural (dairy/beef cattle) sources.

- **Morphology**: The morphology of a water body pertains to the variation in its depth and width, the structure and substrate of its bed, and the structure of the shore. A substantial majority of this type of failure arose from physical modifications, with negative impacts on fish or invertebrates.

- **Hydrology**: The flow and water levels of a water body, which can be adversely affected by abstraction (groundwater and surface water) or impoundments (physical modifications).

- **Nutrients**: High levels of Nitrate or Phosphate. Phosphate accounts for the vast majority of nutrient failures.

- **Sediment**: Relates to siltation, and its adverse impacts on aquatic ecosystems, such as the potential for salmon spawning. Because the sources of the pressure tend to be more diffuse (e.g. agricultural soil compaction), and routine monitoring of sediment itself does not take place, issues over sediment can be difficult to identify (Environment Agency 2013c), and are perhaps under-recorded.

- **Specific pollutants**: Substances that can have a harmful impact on biological quality, and that are discharged in significant quantities. They are listed in Annex VIII of the WFD. Note that the identity of specific pollutants leading to failure pressures was not recorded prior to the development of the EA's Catchment Planning System (CPS), and thus a number of the pressures arising from specific pollutants were classified as 'unknown' (Charlton 2015).

- **Other:** This is a grouping category added by the Project C team, and it includes: Priority substances (Annex X of WFD); Invasive non-native species; pH (high, or lowacidification); Urbanisation (resulting in physical modifications); Drinking water supply (relating to the construction of reservoirs or impoundments for drinking water); Flood protection (physical modification for operational management, or by building structures); Organic pollution (the entry of highly degradable organic material into environmental waters); and Land drainage (physical modification). For approximately 9% of failures, the pressure has not been identified (with an entry of 'Other [not in list]'). The corresponding entries for the sources and sectors of these failures are various, and do not show a particular bias, in terms of category or otherwise.

Appendix Table 2 The pressures identified as the cause of failures in the Environment Agency's 'Reasons for not achieving good' dataset.

Pressure (Level 1)	Share (%)	Pressure (Level 2, if recorded)	Share (%)
DOC (Dissolved Oxygen	2.8		
Ammonia	6.2		
BOD (Biochemical Oxygen	2.8		
Morphology	21.6		
Hydrology	3.7		
Nutrients	20.9	Nitrate	0.4
		Phosphate	20.5
Sediment	12.0		
Specific pollutants	9.8	Unknown	7.5
		Other	0.4
		Zinc	0.7
		Copper	0.7
		Iron	0.6
Other	20.1	Priority substances	3.0
		Invasive non-native species	1.7
		pH	1.4
		Urbanisation	1.2
		Drinking water supply	1.2
		Flood protection	1.0
		Organic pollution	1.0
		Land drainage	0.6
		Other	9.1

Sources

- **Diffuse source**: Leaching of pollutants from the land into water via surface runoff, or interflow.

- **Point source**: Entry of pollutants to the water from a specific site, and often more readily identified. Often sewage discharges.

- **Physical modification**: Physical modifications can be defined as "Manmade changes to the natural habitat, for example poorly designed or redundant flood defences and weirs, and changes to the natural river channels for land drainage and navigation and shellfisheries on estuaries and in coastal waters. These modifications can cause changes to natural flow levels, excessive build up of sediment, and the loss of the habitat that wildlife needs to thrive" (NRW 2014).

- **Natural**: Failures arising from natural conditions. This category covers a wide variety of pre-existing conditions in the water environment that can mean a particular water body may fail WFD criteria in the absence of human activity or a change in that activity. Natural mineralisation may increase copper or zinc levels, or affect the pH. Natural low flows can also result in failure due to morphology criteria.

- **Flow:** As per the table, the majority of 'flow' failures pertain to abstraction (this 3% figure represents surface water abstraction and groundwater abstraction combined). To be cited as a failure, surface water abstraction must be the 'main reason' for low flow; similarly, for groundwater abstraction, groundwater abstraction impacts must represent 'a significant contribution (>50%)' to the low flow. Thus abstraction may contribute towards other failures in this category by less than these required thresholds. Impoundments in which water is stored can also result in flow failures (as can impoundments without storage, although these result in failures less often).

- **Other: This is a grouping category added by the Project C team. Invasive Non-Native Species** comprise the vast majority of the failures in this category (with North American signal crayfish, *Pacifastacus leniusculus*, the most frequently cited INNS).

Appendix Table 3 The sources (Tier 1 and 2) of pressure leading to failures described in the Environment Agency's 'Reasons for not achieving good' dataset.

Source (Tier 1)	Share (%)	Source (Tier 2)	Share (%)
Diffuse source	33.4	Mixed agricultural	8.8
		Dairy/beef field	5.3
		Drainage	5.0
		Arable field	4.4
		Abandoned mine	2.2
		Sewage discharge (diffuse)	1.6
		Farm infrastructure	1.3
		Bank poaching	1.2
		Atmospheric deposition	0.7
		Other	3.1
Point source	30.7	Sewage discharge (continuous)	19.5
		Sewage discharge (intermittent)	4.8
		Unsewered domestic sewage	2.4
		Abandoned mine	1.6
		Industrial/trade discharge (EPR and non-EPR)	1.5
		Other	0.9
Physical modification	25.0	Barriers to fish migration	5.6
		Land drainage	4.5
		Flood protection	4.4
		Urbanisation	3.7
		Impoundments	2.9
		Other	3.9
Natural	4.9	Natural mineralisation	1.6
		Low flows	1.4
		Other	1.8
Flow	4.7	Abstraction	3.0
		Water storage	1.1
		Other	0.6
Other	1.3	Invasive non-native species	1.0

SEPA WFD data tables

Sectors

Appendix Table 4 The sectorial provenance (% share) of SWMIs identified by SEPA.

Industry sector	Share (%)
Renewable electricity production	17.7
Not specified	16.7
Mixed farming	11.2
Sewage disposal	10.6
Water collection, purification and distribution	9.5
Arable farming	7.8
Forestry	5.7
Livestock farming	4.6
Coal mining or quarrying	2.5
Other	13.7

Pressures

Appendix Table 5 The pressures identified as the cause of failures in SEPA's Scottish data

Assessment category	Share (%)	Assessment parameter	Share (%)
Morphology and fish continuity	33.0	Fish passage	13.0
		Multiple pressure	10.5
		Riparian vegetation	3.2
		Single pressure	6.3
		Other	0.0
Water Flow and Water Levels	29.1	Change from natural flow	21.6
		Change in lake outflow	3.5
		Groundwater baseflow depletion	3.6
		Other	0.5
General Water Quality	26.1	Ammonia	2.4
		Dissolved Oxygen	2.2
		pH	2.4
		Phosphorus	18.7
		Other	0.3
Other	11.8	Unknown organics	3.0
		Chemical status	2.5
		Quantitative status	2.5
		Specific pollutants	1.8
		Other	2.1

Sources

Appendix Table 6 The sources of pressure leading to failures described in SEPA's Scottish data.

Pressure type	Share (%)	Industry sector	Share (%)
Morphological alterations	33.0	Not specified	14.0
		Forestry	3.8
		Renewable electricity production	3.7
		Water collection, purification and	3.1
		Mixed farming	2.8

Pressure type	Share (%)	Industry sector	Share (%)
		Other	5.6
Diffuse source pollution	24.0	Mixed farming	7.6
		Livestock farming	3.5
		Arable farming	2.5
		Sewage disposal	2.2
		Not specified	2.1
		Forestry	1.9
		Non-renewable electricity production	1.8
		Mining and quarrying	1.7
		Other	0.8
Abstraction	21.5	Renewable electricity production	7.8
		Arable farming	4.4
		Water collection, purification and	3.5
		Other	5.9
Point source pollution	10.8	Sewage disposal	8.3
		Other	2.5
Flow regulation	10.2	Renewable electricity production	6.3
		Water collection, purification and	2.9
		Other	1.0
Other	0.5	Other	0.5

The overall status of UK water bodies as reported under EU Water Framework Directive (EU 2000, 2008) requirements

Appendix Table 7 The overall status of UK water bodies as reported under EU Water Framework Directive (EU 2000, 2008) requirements (as per Figure 2-1).

River Basin District	Bad	Poor	Moderate	Pass
Anglian	0.6	8.9	52.9	37.6
Dee	0.0	9.6	38.3	52.2
Humber	1.5	12.9	51.4	34.2
Neagh Bann	4.4	22.0	57.3	16.3
North Eastern	3.3	29.3	50.0	17.4
North West	1.5	6.5	38.2	53.8
North Western	0.0	10.1	57.3	32.7
Northumbria	1.7	8.0	23.7	66.6
Scotland	2.9	12.2	20.8	64.0
Severn	1.5	11.8	37.1	49.6
Solway Tweed- England geographic area	0.2	1.4	6.4	92.1
Solway Tweed- Scotland geographic area	3.8	16.4	29.6	50.2
South East	1.1	8.6	41.7	48.5
South West	0.8	6.1	28.0	65.1
Thames	1.5	17.0	42.8	38.7
Western Wales	0.0	5.3	20.1	74.6

Data were derived from: the Environment Agency's data for England and Wales (WFD Cycle 2: predicted 2015 classification data); SEPA's WFD Cycle 2 online consultation tool (2013 data, excludes groundwater); and NIEA 2014 data.

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