Regulating services

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Chapter 14:
Regulating Services

Coordinating Lead Author: Pete Smith
Lead Authors: Mike Ashmore, Helaina Black, Paul Burgess, Chris Evans, Rosemary Hails, Simon G. Potts, Timothy Quine and Amanda Thomson
Contributing Authors: Koos Biesmeijer, Tom Breeze, Mark Broadmeadow, Robert Ferrier, Jim Freer, Jim Hansom, Phil Haygarth, Helen Hesketh, Kevin Hicks, Andrew Johnson, David Kay, William Kunin, Allan Lilly, Linda May, Jane Memmott, Harriet Orr, Roger Pickup, Beth Purse and Geoff Squire

Key Findings*

The regulating services provided by ecosystems are extremely diverse. Their status and trends, drivers and consequences of change, effective management, and knowledge gaps differ greatly There are differences even among indicators within individual regulating services, as can be observed, for example, with the various components of water quality including acidity, pollutants and sediment. The services are, therefore, reported independently, although relevant interactions (particularly between air, soil and water quality regulation) are noted.

Ecosystems regulate climate by: i) providing sources or sinks of greenhouse gases (affecting global warming) and sources of aerosols (affecting temperature and cloud formation); and ii) their physical characteristics which can regulate local and regional climate. The UK has large amounts of carbon 'locked up' in its forests, peatlands and soils (114 megatonnes Carbon (Mt C) in vegetation; 9,838 ± 2,463 Mt C in soils)[1]. Projected changes in emissions (under a 'business as usual' scenario) from the land use and forestry sector over the next ten years could switch this sector from being a net sink of carbon dioxide to a source[c]. The effects of a failure in climate regulation services globally would be particularly pronounced in urban areas, and would exacerbate climate stress for large numbers of people. There are a wide range of sustainable management options to improve climate regulation services, which would also benefit other ecosystem services. Our main knowledge gaps concern the effects of land use management (rather than land use change) on greenhouse gas emissions and removals, and the quantification of the climate regulation provided by urban, coastal and marine ecosystems around the UK.

[1] *well established*
[c] *likely*

The capacity to regulate water, nutrient, pollutant and sediment transfer from the land surface continues to be compromised by soil degradation[2] and contributes to fluvial flood risk. The ability of the soft landforms of the UK coast to regulate erosion (17% currently eroding) and mitigate flood and storm damage is threatened by sea-level rise, changes in the frequency and severity of storms, and low sediment availability[2]. Assessment of the current and future delivery of hazard regulation is limited by our knowledge of coast and upland condition; our understanding of rates and pathways of recovery from degradation; and the need to quantitatively understand the effects of extreme events.

[2] *established but incomplete evidence*

Ecosystems regulate pests and diseases, but this service is under threat. Agricultural intensification, human population growth, accidental introduction of pest and pathogen organisms and land and wildlife management are currently important drivers of disease and pest incidence. Changes in climate are likely to become more important over the next few decades, as recently witnessed for vector-borne diseases. For example, relatively innocuous weeds at the base of the arable food web have declined due to the more frequent use of broader spectrum herbicides, with likely impacts on wider biodiversity[1]. The inadvertent import of fungal plant pathogens via live plant material is arguably one of the most significant loopholes in terms of biosecurity[d]. Understanding how to better manage ecosystems to control pests and pathogens requires detailed longitudinal field studies to describe host-pest and host-pathogen interactions and to understand how these alter in response to environmental changes.

[1] *well established*
[d] *about as likely as not*

Both managed pollinators (honey bees) and wild pollinators (primarily non-managed bees and hoverflies) have been in severe decline for the past 30 years and it is very likely that this trend will continue[1]. Twenty percent of the UK cropped area comprises pollinator-dependent crops, and a high proportion of wild flowering plants depend on insect pollination for reproduction. However, the overall extent of pollination limitation in crops and wildflowers has not been quantified in the UK. The value of pollinators to UK agriculture is conservatively estimated to be £430 million per annum. There are multiple drivers of pollinator loss including loss of semi-natural habitat, the introduction of pathogens, inappropriate use of agrochemicals and climate change[2].

[1] *well established*
[2] *established but incomplete evidence*

* Each Key Finding has been assigned a level of scientific certainty, based on a 4-box model and complemented, where possible, with a likelihood scale. Superscript numbers and letters indicate the uncertainty term assigned to each finding. Full details of each term and how they were assigned are presented in Appendix 14.1.

Noise, or unwanted sound, can have a negative effect on human well-being[1] and certain bird species[2], but can be regulated by ecosystems. Actual spatial measurements of noise are very limited, but national models consistently suggest that noise and visual intrusion has increased[a] as urbanisation, including road traffic, has increased. Ecosystems adjacent to roads (created by tree planting and the use of soil bunds) can reduce some of the effects of road traffic noise[2]. Sounds produced and moderated by ecosystems can also be considered as a cultural service: some natural sounds, such as bird song, are considered positively[2]. In this context, noise can be considered a "disservice".

[1] *well established*
[2] *established but incomplete evidence*
[a] *virtually certain*

Soil quality is linked to almost all other regulating services (e.g. nutrient cycling, biomass production, water quality, climate regulation, pollination, etc.) through the soil's capacity to buffer, filter and transform. Soil quality in all UK NEA Broad Habitats has been degraded by human actions over the last 50 years[1], primarily by atmospheric pollution and inappropriate management practices[b]. Ecosystems are involved in regulating soil quality at all scales. If soil quality is degraded, then soils' capacity to buffer, filter and transform chemical substances is reduced[2]. The trends[2] indicate that recovery from, and remediation of, both diffuse and point source pollution is in progress[c]. There is insufficient and speculative knowledge regarding the recovery of soils under a changing climate; uppermost among these are the competing explanations for the changes in, and vulnerability of, UK's soil carbon stocks and the role of soil in purifying water resources.

[1] *well established*
[b] *very likely*
[2] *established but incomplete evidence*
[c] *likely*

Ecosystems can have positive effects on air quality, primarily through interception, deposition and removal of pollutants. However, if the rate of deposition of pollutants exceeds critical thresholds, there may be adverse effects on a range of other ecosystem services. Emissions to the atmosphere from ecosystems can also directly and indirectly degrade air quality. Although there have been significant improvements in UK air quality over recent decades, current concentrations and deposition rates still exceed thresholds for effects on human health, crop and forest production, and biodiversity over significant areas of the country[1b]. The national improvements in air quality are primarily due to reduced anthropogenic emissions from the transport and energy sectors[1b]. In contrast, the main drivers of changes in the ecosystem service of air quality regulation over recent decades are likely to have been those changes in land use and management which influence deposition and emission of pollutants. It is likely that there are local benefits of tree planting for air quality in urban areas and close to point sources of pollution; these benefits have been quantified for individual air pollutants[2]. However, the overall national benefits of ecosystem regulation for air quality, and for its health and ecological impacts, are very uncertain.

[1] *well established,*
[b] *very likely*
[2] *established but incomplete evidence*

Since the 1980s, water quality has improved in the uplands because lower atmospheric pollution levels in these areas enable terrestrial ecosystems to buffer lakes and streams against acidification and nitrate leaching[1]. In the lowlands, water quality improvements have largely been driven by better control of point source pollution, rather than improved ecosystem regulation of diffuse pollutants[1]. Widespread increases in upland dissolved organic carbon concentrations have had negative consequences for water treatment, but appear linked to soil recovery from acidification[3]. The key regulating service of pollutant dilution by water flow is maximised by land management that increases infiltration rates; this also reduces phosphorous, sediment and faecal pollutant losses via overland flow[1]. However, these ecosystem services are likely to be degraded in the future by more extreme droughts and high flows due to climate change.

[1] *well established*
[3] *competing explanations*

While there are a number of synergies between regulating services (e.g. tree planting can improve air quality, reduce noise and sequester carbon for climate regulation), there are also a number of trade-offs (e.g. improvements in soil buffering of water quality as a result of decreased acid deposition may cause more carbon to be released from upland soils).

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* Each Key Finding has been assigned a level of scientific certainty, based on a 4-box model and complimented, where possible, with a likelihood scale. Superscript numbers indicate the uncertainty term assigned to each finding. Full details of each term and how they were assigned is presented in Appendix 14.1.

14.1 Regulating Services and Human Well-being

A number of regulating services either act as final ecosystem services, such as climate regulation and hazard regulation, or contribute significantly to final ecosystem services, such as water quantity (which is also a provisioning service), detoxification and purification (arising from air, water and soil quality). Other regulating services are primary or intermediate ecosystem services, such as the effects of pollination and pests and diseases on regulating the provision of crops, other plants and livestock.

In this chapter, we focus on eight regulating services which map on to the Millennium Ecosystem Assessment (MA) (2005) regulating services, as supplemented by further ecosystem service research frameworks in the EU and the UK[1]. These are discussed in detail in Chapter 2. The regulating services considered in this assessment are as follows:

Climate regulation is a final ecosystem service. It provides goods and services that regulate climate so that adverse impacts on human well-being and biodiversity are avoided. Ecosystems regulate climate through biogeochemical effects and biophysical effects (MA 2005; **Table 14.1**). Biogeochemical effects operate at the regional or global scale, while biophysical effects operate at the local or regional scale. Local effects in particular assist in the avoidance of climate stress. Climate regulation has strong links with the other regulating services and with provisioning services.

Hazard regulation is a final ecosystem service that provides the following goods and services: coastal protection, erosion protection, flood protection, and to a lesser extent,

the avoidance of climate stress. The regulating ecosystem services associated with individual hazards are defined in **Table 14.2**. This framework of services is used to consider the capacity for regulation of the associated hazards.

Disease and pest regulation is an intermediate ecosystem service which directly affects human health and well-being (in the case of human pests and diseases) and has a potentially large impact on regulating the provision of final ecosystem services, such as crops, other plants and livestock, which deliver food and fibre, amongst other things. This regulating service largely relates to the role of ecosystems in regulating the incidence and spread of insect pests of crops and pathogens of consequence to humans, livestock, crops and ecosystems.

Pollination is a primary or intermediate ecosystem service which has a potentially large impact on regulating the provision of final ecosystem services, such as crops and other plants, which deliver food and fibre. This regulating service largely relates to the role of ecosystems in regulating the incidence and spread of wild and managed insect pollinators. The honeybee is the only species of pollinator that is widely managed; the majority of pollination services are provided by wild bees and other insects. Diversity within wild communities of pollinators provides resilience against environmental change and can be supported through the provision of natural and semi-natural habitats, including agri-environment schemes, which provide a range of flower communities.

Noise regulation is considered as a final regulating ecosystem service in this report. However, the sounds produced and moderated by ecosystems could also be considered as a 'cultural service' as some natural sounds are considered beneficial. Noise, i.e. unwanted sound, can be considered a 'disservice'.

Soil quality regulation is a primary or intermediate regulating service which has a pivotal role in delivering regulating services through the storage and degradation of organic

Table 14.1 Biogeochemical and biophysical mechanisms through which ecosystems regulate climate.

Biogeochemical effects	Sources or sinks of GHGs affect radiative forcing and leads to climate warming
	Sources of aerosols that can reflect or trap solar radiation (warming or cooling effect) and affect cloud formation
Biophysical effects	**Local**
	Provision of shade from heat and UV light
	Provision of shelter from wind and precipitation
	Regulation of temperature
	Regulation of humidity and precipitation
	Regional/global
	Surface albedo—affects radiative forcing and temperature
	Evapotranspiration—affects radiative forcing, cloud formation and precipitation
	Surface roughness—affects winds

Table 14.2 The regulating ecosystem services associated with individual hazards.

Hazard	Ecosystem Service (how ecosystems reduce the hazard)
Mass movements, coastal erosion and flooding	The service is the maintenance of the integrity of landsurfaces (regolith and landforms)
Soil erosion	The service is soil retention on the land surface that is evident in two further services: i) maintenance of 'intact' soil cover while allowing for gradual evolution (on timescales of natural soil formation); ii) maintenance of low suspended sediment loads in fluvial systems
Runoff generation and flooding	The service is water retention and storage and delayed release from the land surface and attenuation of peaks as floodwater passes through river networks

1 http://naturalengland.etraderstores.com/NaturalEnglandShop/R701; www.ecosystemservices.org.uk

matter, mediating exchange of gases to the atmosphere, storing, degrading and transforming materials such as nutrients and contaminants, and regulating the flow of water. This regulating service largely relates to the role of ecosystems in regulating soil quality, and contributes significantly to other final ecosystem services, such as climate regulation, detoxification and purification, as well as the provision of crops and other plants (delivering, for example, food and fibre), trees, standing vegetation and peat.

Air quality regulation is a primary or intermediate regulating service that influences the atmospheric concentrations of air pollutants and their deposition to land and water surfaces. Nationally, the most significant pollutants are particles, ozone, nitrogen oxides, ammonia, and nitrogen and sulphur, the deposition of which can lead to ecosystem acidification and eutrophication. Ecosystems contribute to air quality because they remove pollutants from the atmosphere: gases and particles are deposited to ecosystem (primarily plant) surfaces, and pollutant gases enter leaves through stomata. The extent of this removal depends on a number of factors, including the turbulence of the air above the ecosystem (taller vegetation has a greater efficiency), the duration of leaf cover (evergreen tree species are more effective than deciduous species), and the stomatal aperture of the vegetation (deposition may decrease under drought conditions). However, this deposition or uptake into the leaf can have negative effects on other ecosystem services if critical thresholds are exceeded. The deposited or absorbed pollutants may be detoxified (e.g. ozone), assimilated by plants or microbial activity (e.g. nitrogen oxides and ammonia), or simply accumulate within the ecosystem (e.g. lead and other metals). The detoxification process may occur at the expense of utilising resources for plant growth, therefore, decreasing production, while assimilation may differentially increase the growth of different plant species, changing competition and leading to a loss of less competitive species. Ecosystems can also have negative effects on air quality due to emissions to the atmosphere, which increase air concentrations either directly or indirectly through chemical reactions in the atmosphere. Therefore, ecosystems contribute to final ecosystem services such as detoxification and purification. Because this regulatory service affects the concentrations and deposition of air pollutants, it may, in turn, affect the provision of final ecosystem services such as the production of crops and the growth and biomass of trees.

Water quality regulation is a primary or intermediate regulating service determined primarily by catchment processes. It is, therefore, intimately linked to other ecosystem services addressed in this section (e.g. soil and air quality, climate regulation) and elsewhere in the report (e.g. nutrient cycling (Chapter 13)). Key ecosystem processes regulating water quality include plant and microbial nutrient uptake, pollutant sequestration in soil and sediment organic matter, breakdown of organic pollutants, acidity buffering and denitrification. These processes contribute to final ecosystem services including detoxification and purification, drinking water and fisheries provision, and recreation (e.g. bathing waters).

Numerous cultural services (Chapter 16) are also supported by ensuring good water, air and soil quality, including physical health, mental health, ecological knowledge, cultural heritage and mediated natures, and aesthetic and inspirational.

In the following sections, we examine each regulating service in turn and describe: i) the condition, status and trends; ii) drivers of change; iii) consequences of change; iv) knowledge gaps, and v) sustainable management for each regulating service. We conclude with a section summarising the key synergies and trade-offs between different regulating services, and between key regulating services and other ecosystem services. And we consider how regulating services might be managed to promote synergies and minimise trade-offs.

14.2 Climate Regulation

14.2.1 Trends in Climate Regulating Services

The biosphere, and the ecosystems within it, regulate climate by controlling the flux of GHGs (principally carbon dioxide, but also methane and nitrous oxide), sources of aerosols and the transfer of heat, moisture and momentum (Burroughs 2001; Houghton 2004; Beaumont *et al.* 2007; Bonan 2008; Fowler *et al.* 2009). Ecosystems also regulate microclimate, dampening temperature extremes and providing shade and shelter. The processes involved in climate regulation include:

- Photosynthesis which is the fundamental process affecting levels of carbon dioxide in the atmosphere. Through negative feedback, the biosphere can also act as a temporary sink for additional carbon dioxide entering the atmosphere from fossil fuel combustion.
- Marine organisms acting as a carbon sink in the ocean and facilitating carbon burial in seabed sediments.
- Evapotranspiration from soil and plants which controls the amount of water vapour entering the atmosphere, regulating cloud formation and the radiative properties of the atmosphere.
- The albedo values of different land surfaces (i.e. the proportion of incoming solar radiation that is reflected from the Earth's surface). A change in albedo, through a change in vegetation for example, can have a cooling or heating effect on the surface climate and may affect precipitation.
- The production of aerosols by ecosystems from soil erosion or vegetation, and the regulation of aerosols through vegetation scavenging, which affects radiative heating of the atmosphere and surface albedo (Bonan 2008; Section 14.8).

Nearly 10 billion tonnes of carbon is stored in UK soils (Dawson & Smith 2007): over half of this occurs in organic soils (Smith *et al.* 2007) predominantly in Scotland and Wales (**Figure 14.1**). Soil carbon densities are greatest under

semi-natural habitats (Mountains, Moorlands and Heaths, Semi-natural Grasslands, wetlands) and Woodland habitats (including those with peat soils) and lower under Enclosed Farmland and Urban habitats (**Figure 14.2**). There are significant biomass carbon stocks in the forest plantations planted during the 20th Century (150 million tonnes). Soil carbon densities and storage can be high in Coastal Margin habitats, but they are limited in extent. There will be considerable carbon stores in the Marine habitats around the UK but these have yet to be quantified. The marine ecosystem balances and maintains the chemical composition of the atmosphere and oceans (Beaumont *et al.* 2008).

The management of ecosystems can result in emissions of GHGs through the cultivation of previously undisturbed soils, peat extraction or the production of methane by ruminant livestock, amongst other things. Or it can counter emissions through removals, for example, via forest planting and the removal of land from cultivation (**Box 14.1**). Ecosystem products such as timber and biomass crops, can also store carbon or replace other products with higher emission costs. Peat bogs are the only major soil source of methane in the UK, but the soil sink for methane is insignificant in this country (Fowler *et al.* 2008).

Setting aside existing long-term carbon stores, UK ecosystems have produced an increasing net sink of carbon dioxide since 1990, as reported in the Land Use, Land Use Change and Forestry (LULUCF) sector of the UK Greenhouse Gas (GHG) Emissions Inventory (Jackson *et al.* 2009; MacCarthy *et al.* 2010).This sector is projected to become a net source by 2014 under a 'Business As Usual' scenario (Thomson *et al.* 2010) as afforestation rates have slowed and large areas of conifer plantations are reaching maturity, leading to a declining rate of net carbon uptake by UK forests. The Cropland and Settlement land use categories are net sources, while the Forest Land and Grassland categories are net sinks. Among the countries of the UK, England is a net source of carbon dioxide (although this has diminished from 1990 to 2007), whereas Scotland, Wales and Northern Ireland are net sinks. Methane emissions from ruminant livestock are a major source of GHGs from agriculture, but have reduced by 13% since 1990 in line with livestock numbers (Jackson *et al.* 2009; MacCarthy *et al.* 2010). Nitrous oxide emissions from agricultural soils arise from nitrogen application, leaching of fertiliser and manure to ground and surface water, ploughing in crop residues, atmospheric deposition of ammonia and nitrogen oxides, cultivation of highly organic soils and biological fixation in improved grass. These sources are the largest contributors to nitrous oxide emissions across the UK, from 64% in England to 81% in Northern Ireland. They contribute 3–9% of the total 2007 GHG emissions in each country. But overall, nitrous oxide emissions have decreased by 23% since 1990, a decline driven by a fall in synthetic fertiliser application and reduced livestock numbers. Enclosed agricultural land has the greatest nitrous oxide emissions, with woodlands, mountains, moorlands and heaths contributing less than 5% to the total soil emissions (Sozanska *et al.* 2002).

The status and trends in climate regulation by habitat are summarised in **Table 14.3**. More detail is given in Chapters 5 to 12.

14.2.2 Drivers of Change

The direct drivers of change in climate regulation and their effects are summarised in **Table 14.4**. Full descriptions of these drivers and the related indirect drivers of change are given in Chapter 3. Land use drivers are the most immediately important in the ecosystem context, but over
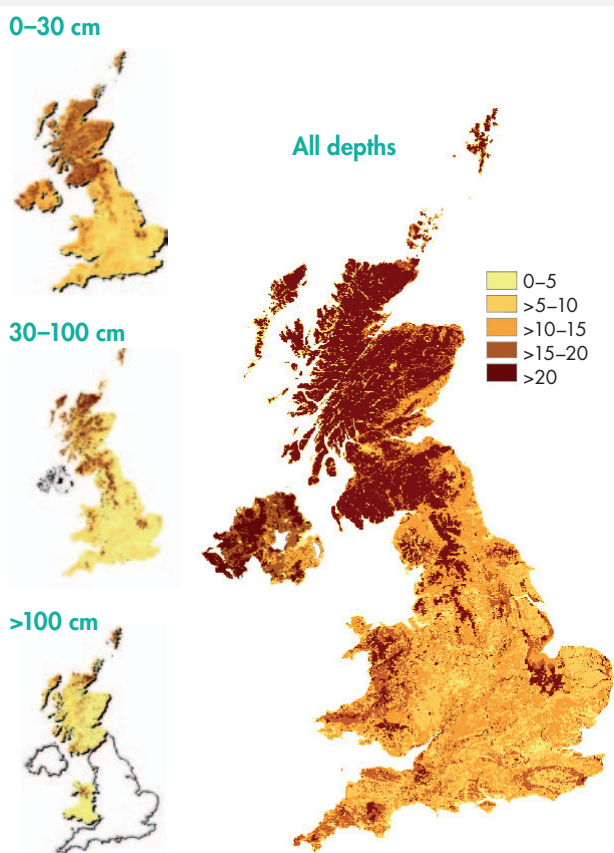
Figure 14.1 Density (kg per m²) of soil carbon in the UK. Source: Bradley *et al.* (2005). Copyright © 2005 British Society of Soil Science. Reproduced with permission of Blackwell Publishing Ltd.
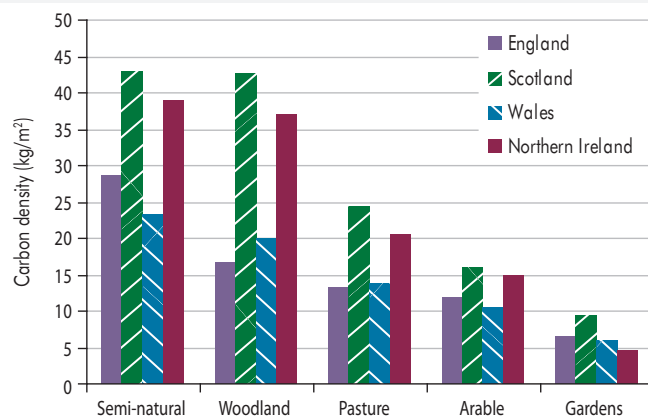
Figure 14.2 Soil carbon density 0–100 cm under different land use types in the UK. Source: data from Bradley *et al.* (2005) and Smith *et al.* (2007). Copyright © 2005 British Society of Soil Science. Reproduced with permission of Blackwell Publishing Ltd.

Box 14.1 The UK greenhouse gas inventory.

The UK produces annual inventories and projections of GHG emissions. These help monitor progress on reducing emissions for international and national targets, e.g. the Kyoto Protocol of the UN Framework Convention on Climate Change and targets in the UK Climate Change Act. The national inventory (MacCarthy *et al.* 2010) has six sectors:

- Energy
- Industrial Processes
- Solvent and Other Product Use
- Agriculture
- Land Use, Land Use Change and Forestry (LULUCF)
- Waste

It follows the methods developed by the Intergovernmental Panel on Climate Change. Emissions are reported for 1990 to the present, and projections to 2020.

The LULUCF sector reports carbon stock changes of carbon dioxide (can be emissions or removals) and emissions of methane and nitrous oxide from land use activities (Figure 1 and Figure 2). The broad categories for reporting are:

- Forestry
- Cropland
- Grassland
- Settlements

Wetland areas are included under Grassland. Emission estimates are made using models driven by activity data from government agencies, such as the Forestry Commission.

More information is available at: www.edinburgh.ceh.ac.uk/ukcarbon

Figure 1 UK LULUCF emissions and removals in Gigagrams of carbon dioxide (Gg CO$_2$) from 1990 to 2008. Source: reproduced from MacCarthy et al. (2010) © NERC (CEH).

Figure 2 Carbon stock changes due to Article 3.3 Afforestation in the UK from 1990 to 2007. Carbon stock changes modelled on planting data from the Forestry Commission. Source: reproduced from Dyson *et al.* (2009).
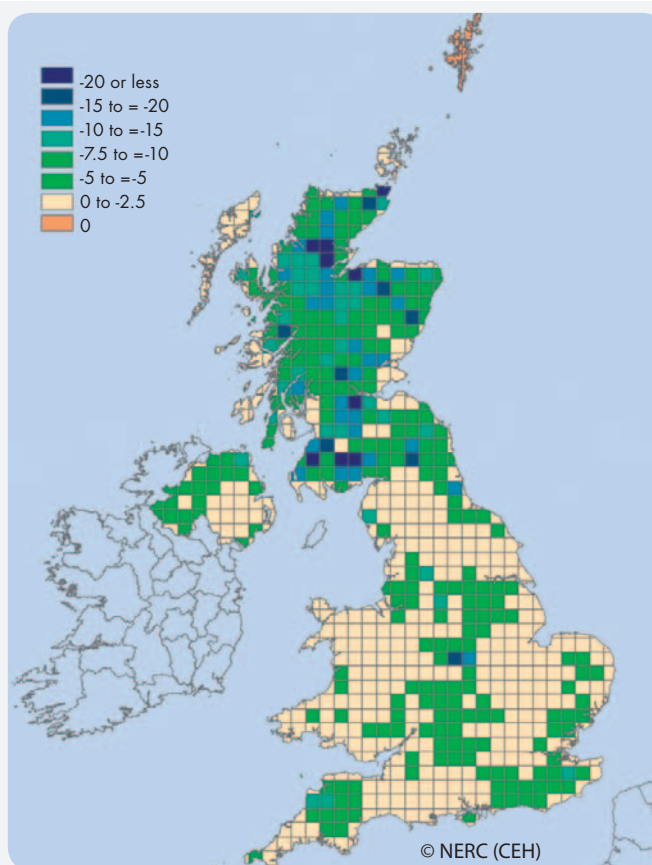
the longer term, climate change will also feedback to climate regulation services (Schröter *et al.* 2005).

14.2.3 Consequences of Change

The increased levels of GHGs in the atmosphere as a result of human action (burning fossil fuel and land use change) during the past 200 years are resulting in climate change on a global scale (IPCC 2007a). Global climate change threatens all human populations, with those most affected often having contributed least to the problem (IPCC 2007b). Ecosystem services that regulate GHGs can affect the whole atmosphere, not just that above a specific region, so national policies have global consequences.

The effects of climate regulation are particularly pronounced in urban areas. A reduction in climate regulation services could exacerbate climate stress for large numbers of people, reducing well-being and increasing death rates (for example, through higher summer temperatures). Urban areas in southern England are probably at greatest risk.

Changes in temperature and moisture regimes will shift ecosystem boundaries and have consequences for biodiversity. This could lead to potential changes in ecosystem composition as species that are operating at the edges of their climatic range lose or gain ground. Under a warmer climate this could particularly affect biodiversity in the Mountain, Moorlands and Heaths (Chapter 5),

Table 14.3 The status and trends in climate regulation by UK NEA Broad Habitat. The status includes UK NEA uncertainty terms.

UK NEA Broad Habitat	Status		Trends in climate regulation
Mountains, Moorlands and Heaths	**+**	Positive (extensive carbon stores, few other GHG emissions) (*established but incomplete, very likely*).	Losses of carbon storage potential in past (particularly in England) but this trend is potentially reversing with increased focus on peatland restoration.
Semi-natural Grasslands	**– / +**	Mixed. Significant carbon stores under acid grasslands (*established but incomplete, likely*), but also GHG emissions from livestock, nitrogen fertilisation and liming.	Probably stable. Loss of carbon storage capacity has occurred in the past but not recently (according to Carey *et al.* 2007).
Enclosed Farmland	**–**	Overall negative impact on with high emissions of GHGs from agricultural management and low carbon stocks in soils (*well established, very likely*).	Continuing loss of carbon storage capacity in soils. Loss of shelter from degradation of hedgerows and shelterbelts. Reduced methane emissions due to reductions in livestock numbers.
Woodlands	**+**	Positive (extensive stores and sequestration of carbon in vegetation and soils), provision of shade and shelter (*well established, very likely*).	Increasing trend over time due to extensive tree planting in past 50 years.
Freshwaters—Openwaters, Wetlands and Floodplains	**+**	Positive (extensive carbon stores) (*well established, very likely*). Also affect regional climate (cooling effect, reduced evaporation, modifies cloud cover).	Losses of carbon storage potential in past but this trend is potentially reversing with increased focus on wetland restoration.
Urban	**–**	Overall negative impact: the urban heat island effect increases climate stress, but there can be significant quantities of carbon stored in vegetation and soils in urban areas (*competing explanations*). Urban greenspace can have high levels of GHG emissions due to the use of fertilisers and drainage.	Negative trend over time.
Coastal Margins	**+**	Positive, can have high rates of carbon sequestration but total area is small (*established but incomplete*).	Decrease in area has led to loss of carbon storage in the past but managed realignment may increase area of saltmarshes (and carbon storage capacity).
Marine	**+**	Positive (extensive carbon stores and uptake of atmospheric carbon dioxide), also affect regional climate (weather patterns).	Continuing uptake of carbon dioxide but ocean acidification may lead to functional changes in the future.

Table 14.4 Direct drivers of change in climate regulation and their effects.

Driver category	Driver of change	Effect on climate regulation	UK NEA Broad Habitats most affected
Habitat change: Land and sea use	Productive area: expansion, conversion, abandonment (agriculture, forestry)	Affects carbon sinks and existing stores, GHG emissions, albedo and evapotranspiration, shade and shelter	Semi-natural Grassland Enclosed Farmland Woodlands Freshwaters Marine
	Mineral and aggregate extraction (peat)	Affects soil carbon stores, GHG emissions	Mountains, Moorlands and Heaths Freshwaters
	Urbanisation and artificial sealed surfaces	Affects soil carbon stores, albedo, shade, shelter, local temperatures and humidity	Urban
Pollution and nutrient enrichment	Pollution emissions and deposition	Affects aerosol sources (soot)	Urban
	Nutrient and chemical inputs	Affects GHG emissions	Enclosed Farmland
Harvest levels/resource consumption	Livestock stocking rates	Affects GHG emissions	Semi-natural Grasslands Enclosed Farmland
Climate variability and change	Temperature and precipitation	Affects existing carbon stores, evapotranspiration	All
	Carbon dioxide and ocean acidification	Affects existing carbon stores, GHG emissions, aerosol sources	Marine
	Sea-level change	Affects existing carbon stores	Coastal Margins

and Coastal Margin (Chapter 11) habitats. Rising sea temperatures are already affecting species distributions in the marine environment (Chapter 12).

Threats to the integrity of the UK's carbon stores (particularly deep peats) have the potential to release massive amounts of carbon dioxide into the atmosphere—the carbon equivalent of five years' worth of England's annual carbon dioxide emissions is stored in English peatlands alone (584 million tonnes of carbon) (Natural England 2010). Habitat restoration of degraded ecosystems with deep peats (Chapter 5) could also enhance their carbon sink potential and counter other emissions. However, in some cases, restoration achieved through raising water levels in deep peats may result in methane release through other mechanisms, resulting in a net GHG source in the short-term (10–30 years), although the long-term balance will be positive due to increased carbon sequestration.

Changes in carbon sinks may affect other ecosystem services: for example, land use change may initiate erosion, affecting soil, water or air quality and/or increasing vulnerability to hazards like flooding (although these are avoidable if change is sensitively undertaken). Conversion of semi-natural habitats to agricultural land uses can result in losses of both carbon stocks and biodiversity: a lose-lose situation. Some changes which have produced a net carbon sink, such as the widespread planting of non-native conifers, have had adverse effects on biodiversity (a win-lose situation) with area lost from Mountains, Moorlands and Heaths and Semi-natural Grassland habitats as well as broadleaved woodland.

14.2.4 Options for Sustainable Management

There are already efforts in place to encourage the biogeochemical aspects of climate regulation that will reduce GHG emissions and enhance GHG sinks. The UK has international commitments under the terms of the Kyoto Protocol, as well as domestic targets for reducing emissions, and each of the devolved administrations is in the process of developing national targets and climate change strategies (HM Government 2009; DECC 2009; Scottish Government 2009; WAG in press). Responsibility for agriculture and forestry is devolved to individual countries where sector emission reduction targets are still being developed; for example, English farmers have an emission reduction target of 6% below current predictions by 2020 under the Low Carbon Transition Plan (HM Government 2009; CCTF 2010). Management options for UK forestry are reviewed in Read *et al.* (2009), and each country has developed its own forestry strategy: the Strategy for England's Trees, Woods and Forests, the Scottish Forestry Strategy, Woodlands for Wales, and Northern Ireland's Department of Agriculture and Rural Development Forest Service Strategy for Sustainability and Growth. Sustainable management affecting sources or sinks of GHGs include the maintenance of existing carbon sinks, and the reduction of emissions or the increase in carbon storage as a result of human activities (**Table 14.5**).

Implementation of sustainable management strategies should include a complete assessment of all potential GHG losses and savings over the short- and long-term (Nayak *et al.* 2008; Six *et al.* 2004) and the repercussions for other

ecosystem services. For example, wetland restoration through plantation removal may actually result in net emissions in the short-term if non-carbon dioxide losses and loss of carbon sequestration in forest biomass are taken into account, but a net sink over the long-term as the carbon sink capacity of the peat is restored. There are also biodiversity benefits to such restoration. However, it is important to ensure that reducing emissions in one instance does not directly result in greater emissions globally through displacement abroad ('leakage').

Sustainable management can also enhance local climate regulation, particularly in urban areas where it will have direct benefits to most of the human population. Tree planting and the creation or restoration of urban greenspaces can increase shade and shelter, reduce the urban heat island effect, improve water retention in soils and reduce runoff (reducing flood hazard risks), and reduce pollution loads (through scavenging by vegetation). Such management also benefits cultural services by making urban areas more pleasant places to live and work, and provisioning services, for example, through allotments and improved water management. Hard impermeable surfaces sealing the soil, such as asphalt car parks, could be replaced with semi-permeable surfaces, such as honeycomb structures, which retain soil functions including heat absorption and water retention. The continued planting of farm woodland will increase the provision of shade and shelter for livestock and other animals in rural areas, reducing climate stress from higher temperatures and more frequent storms. There are also indications that the planting of shelterbelts on upland grassland can enhance carbon storage (Marshall *et al.* 2009; Chapter 6).

14.2.5 Knowledge Gaps

The biogeochemical aspects of climate regulation (i.e. fluxes of GHGs and sources of aerosols) are widely studied and work is underway to address knowledge gaps in most cases. There are a number of approaches to estimating fluxes of GHGs (Smith *et al.* 2008). While it is possible to measure GHG fluxes directly at regional and national scales by sensors on tall towers, aircraft (Fowler *et al.* 2008) and satellites (e.g. SCIAMACHY), the techniques to apportion fluxes between sources are still being developed (Bousquet *et al.* 2006). At the ecosystem level there are large natural fluxes of carbon dioxide, methane and nitrous oxide, both to and from the atmosphere, which can be highly variable over time (Randerson *et al.* 1997). Assessing the potential of natural carbon sinks or the contribution of human-induced fluxes (from land use practices) to total emissions requires the use of process-based models. There are knowledge gaps in both the information used to parameterise such models and our understanding of the processes underpinning them (Smith *et al.* 2008).

There is a body of knowledge on the carbon stored in the uppermost layers of soils across the UK (Bellamy *et al.* 2005; Bradley *et al.* 2005; Emmett *et al.* 2010) and, in recent years, there has been a focus on the carbon stores in peat soils in particular (Smith *et al.* 2007). Carbon storage in urban areas is less studied (even though green space may make up more than 40% of settlement area) and little is known about

Table 14.5 Sustainable management affecting sources or sinks of greenhouse gases include the maintenance of existing carbon sinks (M), and the reduction of emissions (R) or the increase in carbon storage (I) as a result of human activities.

Management option	Effect	UK NEA Broad Habitats affected	Effects on other ecosystem services
Enhanced protection of soils with high carbon contents that are at risk of degradation through management, land use conversion or erosion.	M	Mountains, Moorlands and Heaths Semi-natural Grassland Enclosed Farmland Freshwaters Coastal Margins	Soil quality (+) Hazard regulation (+)
Restoration of the water table in deep peats through the reduction or removal of drains and plantations.	M, R, I	Mountains, Moorlands and Heaths Woodlands Freshwaters	Water quality (+)
Conversion of organic soils under agricultural cultivation to natural vegetation cover.	M, I	Enclosed Farmland	Provisioning services (-) Pollination (+?)
Reduction of soil disturbance through minimal tillage practices (M, R), and the promotion of perennial cropping (e.g. biomass crops).	I, R	Enclosed Farmland	Soil quality (+)
Return more agricultural residues to the soil.	I	Enclosed Farmland	Soil quality (+)
Improved fertiliser practices to reduce the total applied and minimise losses.	R	Enclosed Farmland	Water quality (+) Air quality (+)
Modify livestock diet and housing, convert animal manures into energy.	R	Enclosed Farmland	Water quality (+) Air quality (+)
Managed coastal realignment (conversion to saltmarsh will reduce overall area losses and possibly enhance carbon sequestration).	M	Coastal Margins Marine	Hazard regulation (+) Water quality (+) Supporting services (+) Cultural services (+) Provisional services (-)
Shifting forest management to longer rotations or continuous cover practices.	M, I over long-term	Woodlands	Provisioning services (+,-) Cultural services (+)
Take opportunities to improve forest product utilisation and management practices that deliver wood fuel and material substitution.	R	Woodlands	Provisioning services (+)
Reduction of urban development on greenfield sites, particularly woodland	R	Enclosed Farmland Woodlands Freshwaters	Provisioning services (+) Soil quality (+)

carbon stores and GHG fluxes in fluvial and coastal habitats in a UK context.

There is less detailed understanding of the impact of different land management practices (which do not result in land use change) on total soil carbon and, therefore, potential emissions and removals. Some of these practices, for example minimum tillage, are thought to have the potential to sequester soil carbon, although this has been challenged by Baker *et al.* (2007). Not enough is known about their potential in a UK context or the permanence of such carbon sequestration (Ogle *et al.* 2005; Manley *et al.* 2005). There is a need for specific measurement and monitoring of emissions from land under such practices. When agricultural policy encourages such practices (see ClimSoil, 2008, for an analysis of European policies affecting soil carbon), especially for their potential to ameliorate climate change, there should be monitoring of actual take-up. The contribution of improved agricultural management cannot be shown in emissions inventories if there is no data on where and when it is happening. There is also a requirement for improved country-specific GHG emission factors for agricultural activities, which will be addressed by the Agricultural Inventory Research Platform recently funded by the Department for Environment, Food and Rural Affairs (Defra) and the Devolved Administrations.

Mountain, Moorland and Heath habitats (Chapter 5), in addition to their existing carbon stores, can either be sources or sinks of carbon dioxide and methane depending on location, management, prevailing climate and season. Information on GHG fluxes from deep peats is, at present, too patchy to make consistent, robust calculations at the UK scale. Understanding of the effect of peat habitat restoration on GHG fluxes could also be improved (Baird *et al.* 2009), and is currently the focus of ongoing research projects funded by Natural England, the Forestry Commission and Defra.

While there is sufficient knowledge of nitrous oxide emissions from agricultural practices to make estimates at a national scale (MacCarthy *et al.* 2010), there are still serious knowledge gaps about emissions from soil under different land uses and under different management practices, largely

due to the temporal and spatial heterogeneity of nitrous oxide emissions (Reis *et al.* 2009).

There is also a need for improved modelling of long-term future scenarios and integration of emissions models with socio-economic models. Land use policy aimed at reducing emissions or conserving carbon sinks does not exist in a vacuum and may have wider implications; for example, returning cultivated land to semi-natural habitats might reduce emissions in the LULUCF sector but also increase dependence on imported food and increase transport emissions. The impact of climate change on land-based carbon stores and sinks, particularly soils, is also an issue of concern.

The biophysical effects of climate regulation at the regional scale are poorly quantified: how do changes in habitat and vegetation structure (e.g. plantation versus open-structured forest) affect regional albedo and exchanges of energy with the atmosphere driving temperatures and rainfall? Will this work to exacerbate or mitigate the effect of global climate change in the UK?

14.3 Hazard Regulation

14.3.1 Trends in Hazard Regulating Services

14.3.1.1 Maintenance of the integrity of landforms

Rates of geomorphic change in the UK's landscape are typically relatively modest and the integrity of landforms is rarely threatened. However, there are two significant exceptions.

Firstly, the beaches, dunes and saltmarshes of the UK's coastline (Chapter 11) are wasting assets that have a declining capacity to provide hinterland protection from erosion and flooding, particularly in the south and east (Pye & French 1993). It is estimated that 17% of the UK's coast (30% in England; 23% in Wales; 20% in Northern Ireland; 12% in Scotland) is experiencing erosion (Masselink & Russell 2008). Furthermore, erosion of these areas represents a significant loss of important habitat. The exceptions are locations such as river exits and estuaries where coastal sediment is locally available, but even here, the erosion and flooding hazard is increasing, compounded by a rise in fluvial flooding. Widespread frontal erosional loss of estuarine saltmarsh and habitat is the norm, particularly in the south. Similarly, in dune systems, the proportion of embryo dune and new dune slack is minimal as a result of declining volumes of sand being fed in from fronting beaches.

Secondly, in the uplands, the last half century has seen increased *reporting* and *impact* of landslides, debris flows and peat-slides. These events are believed to be driven by hydrology, although the precise mechanisms are not well understood (Ballantyne 2004; Warburton *et al.* 2004). Most landslides are linked to intense rainfall events affecting steep slopes with shallow regolith (soil and unconsolidated rock) cover, and most peat-slides are associated with convective summer thunderstorms, rapid snowmelt or intense winter rainfall. Therefore, if climate change results in an increased frequency of extreme rainfall events, then the hazards of mass movements and associated flooding are likely to escalate.

14.3.1.2 Maintenance of soil cover

In the absence of human activity, most of the land surface of the UK would be characterised by full vegetation cover. This would be expected to result in minimal threat to the maintenance of soil cover and low suspended sediment concentrations. However, the capacity for maintenance of soil cover has been reduced by human activity over millennia and, more significantly, over the last 50 years (evidenced by sediment accumulation in dry valleys and lakes). UK uplands are considered to be in a poor condition (Sites of Special Scientific Interest (SSSI) assessments; **Table 14.6; Table 14.7**) with large areas of bare organic soils (McHugh *et al.* 2002), gullying in peatlands (Evans & Warburton 2005) and high sediment export (Holden *et al.* 2007a). The increasing loss of carbon from upland peat soils in the forms of dissolved and particulate organic carbon (DOC and POC, respectively) has been identified (Worrall *et al.* 2003, 2007), the cause of which may include drought frequency or local management and recovery from acidification (DOC only), although no consensus has been achieved (Worrall *et al.* 2007; Orr *et al.* 2008). Reversal of these trends without deliberate

Table 14.6 Soil erosion in upland England and Wales, based on field measurements in 1999 at 399 circular sites each of radius 50 m. Source: adapted from McHugh *et al.* (2002). Copyright © 2002 John Wiley & Sons Ltd. Reproduced with permission of Blackwell Publishing Ltd.

	Volume (m³)	Area (m²)		
	All sites with evidence of erosion (206 sites)	All sites with evidence of erosion (206 sites)	Bare soil exposed (198 sites)	Eroded area revegetated (85 sites)
Total	88,962	77,212	39,186	38,026
Site mean	432	375 (4.8%)	198 (2.5%)	447 (5.7%)
Range	0.04–1,1750	0.16–7,854 (0–100%)	0.02–4,712 (0–60%)	0.1–7,124 (0–91%)

Table 14.7 Soil erosion in upland Scotland. Derived from 1988–1999 aerial photography by Grieve *et al.* (1995) based on 144 tiles (each 5x5 km). Copyright © 1995 John Wiley & Sons Ltd. Reproduced with permission of Blackwell Publishing Ltd.

Erosion type	Percentage of total sampled area	Ranges amongst 16 regions (% of sampled area)
Eroded peat	6.02	0.47–20.43
Gullied area	4.69	0.00–15.40
Debris flow/cone	0.61	0.00–6.80
Landslide	0.08	0.00–0.35
Sheet erosion	0.61	0.00–2.71

	Density (m/km)	Ranges amongst 16 regions (m/km²)
Linear gullies	80	0–236
Footpaths	5	0–72
Vehicle tracks	40	0–183

management is improbable; however, measures have been undertaken to improve condition such as reseeding, blocking grips (artificial drains), reducing grazing and managing burning. The effectiveness of these measures is as yet uncertain, and their full effect may take years to be realised (Holden *et al.* 2007b; Orr *et al.* 2008; Bonn *et al.* 2009). In uplands, semi-natural grassland and pasture there has been some evidence of an increased frequency of localised loss of vegetation cover and exposure of subsoil. Such losses have been attributed to recreational impacts, such as footpath erosion, and high grazing density (**Table 14.7**). In contrast, on some lowland heathland, reduced grazing pressure and nitrogen deposition can result in successional change that is considered undesirable with respect to biodiversity, but the identification of optimum management has proved elusive (Newton *et al.* 2009). Furthermore, some mobility in dune systems is necessary for both coastal defence and habitat diversity; one exception being machair where centuries of seaweed application has promoted deep soil development and a distinctive habitat.

On arable land, water, wind and tillage erosion have accelerated over the last 50 years, but the quantification of these processes at a regional or national scale remains a challenge (**Table 14.8**).

Water erosion is increasingly being observed and local rates of water erosion have also risen significantly (Boardman & Evans 2006).These fluxes have been attributed to changes in the management of agricultural land leading to prolonged surface exposure to erosive agents (Section 14.3.2). Accelerated water erosion occasionally results in very high suspended sediment loads ('muddy flows');indeed, there is some suggestion that the frequency of muddy flows has increased and is likely to continue to do so if the frequency of extreme rainfall events rises in response to climate change. However, evidence for a more general increase in suspended sediment concentrations is less clear. This may reflect the high capacity for storage of sediment within agricultural landscapes between the site of erosion and the fluvial system.

Wind erosion becoming a more common problem, especially where high organic content soils in areas with a high potential for desiccation are subject to cultivation (for example, East Anglia).

Advances in the measurement of soil redistribution have revealed that tillage erosion (the net differential movement of soil by tillage implements) has been responsible for rates of erosion and deposition in fields equal to or higher in magnitude than those caused by water erosion. Tillage erosion is also extremely common on sloping agricultural land (Van Oost *et al.* 2006; Van Oost *et al.* 2007, 2009). This is resulting in the development of high spatial variability in soils on sloping agricultural land, with shallow, impoverished soils on spurs and shoulders, and deep soils in hollows (Quine & Zhang 2002; Quine & Van Oost 2007).

14.3.1.3. Water retention, storage and delayed release (flood risk regulation)

The changes to the landscape that promote water erosion (discussed in Section 14.3.1.2) also promote lower water retention and storage, and a reduction in the mediation of water release. This has the consequence of promoting more rapid runoff and a greater potential for flooding. Furthermore, increases in soil surface compaction and a concomitant reduction in infiltration on grasslands have been attributed to changes to the intensity of grazing. These are associated with small increases in sediment yield and more significant increases in runoff and the rate of pollutant and nutrient transfer from the land surface. Assessing

Table 14.8 Soil erosion on arable land in England and Wales. There is a significant difference between erosion rates derived using caesium-137 (^{137}Cs), which represents all erosion processes, and landscape-scale estimates of water erosion rates. The simulated rates of tillage erosion are of a similar magnitude to the ^{137}Cs-derived rates and it is probable that the difference between the ^{137}Cs-derived rates and typical water erosion rates is due to the contribution to total erosion and deposition made by tillage erosion. This is consistent with analysis of the relative contributions of water and tillage erosion to total erosion in individual detailed field-scale studies (Quine & Zhang 2002).

Erosion process	Method	Area sampled or analysed (soil association code)	Erosion rate (tonnes/ha/year)				Source
			Median gross	Mean gross	Mean gross eroded area	Net	
Rill erosion	Aerial/field survey	1,700 fields					Boardman & Evans (2006)
		92 fields, 2 counties (343)	0.4–0.6	1.0–1.1			
		112 fields, 2 counties (411)	0.3–0.7	0.6–1.4			
		212 fields, 3 counties (541)	0.4–2.1	1.1–4.8			
		516 field, 4 counties (551)	0.5–1.1	1.1–3.2			
		348 fields, 5 counties (571,2)	0.7–1.6	1.3–5.5			
		45 fields, 1 county (581)	0.2	0.9			
Tillage erosion	Simulation	Arable area England & Wales		3.7	7.4		Quine *et al.* (2006)
Root crops	Simulation	Root crop fields		1		0.8–2	Quine *et al.* (2006)
All	^{137}Cs	248 fields	6.6	8.4	10.6	6.7	Walling & Zhang (in press)

changes in land use, Carroll *et al.* (2004) found that the introduction of tree shelterbelts in the Pontbren catchment in mid-Wales increased the overall infiltration capacities by up to 60 times, which implies that rapid surface runoff could be reduced. The effects of such interventions on the overall flood magnitudes in the catchment were evaluated by Wheater *et al.* (2008), who concluded that interventions in land management in the Pontbren significantly reduced peak runoff. In the uplands, changes in flood frequency have been attributed to the creation of rapid flow pathways through the use of grips to drain moorland (Longfield & Macklin 1999; Evans *et al.* 2004, 2008). Recognition of this has led to the management changes outlined in Section 14.3.1.2. Changes in flood frequency and magnitude have been analysed with respect to climate change, but a clear link has not been identified. However, Robson (2002) has suggested that trends towards protracted high flows over the last 50 years could be accounted for by climatic variation. Robson (2002) found that, for the UK, there were clear flood-rich and flood-poor periods, but extremes were seldom shown to be increasing over the last 100 years (Wilby *et al.* 2008). Possible increases in winter precipitation associated with climate change may lead to an increase in flooding (Hulme & Dessai 2008).

14.3.1.4 Climate change trends

Climate change might lead to an intensification of the hydrological cycle and an increase in floods in many parts of the world (Huntington 2006). In the UK, for example, a possible increase in winter precipitation may lead to an increase in flooding (Hulme & Dessai 2008). Growing confidence surrounds the prediction of mean and seasonal changes in rainfall total. Nevertheless, there is significant uncertainty associated with the identification of an increase in frequency of extreme rainfall events (which drive flooding and erosion episodes) in both observed historical records and climate change projections. Wilby *et al.* (2008) discuss many of the issues at length, highlighting various studies, the evidence used, and their conclusions. In summary, evidence of increases in extremes for some studies may have been the result of shorter time periods incorrectly characterising the postulated current UK and European flood-rich period. Projections of extremes, therefore, are still poorly understood, and a greater emphasis needs to be placed on the capability of climate models to be fit for purpose to estimate changes to some types of extreme events.

14.3.2 Drivers of Change

14.3.2.1 Maintenance of stable land surfaces

The drivers of landslides, mass movements and fluvial flooding are those that permit hydrological thresholds to be exceeded. These may include:

- Changed frequency of convective summer thunderstorms, rapid snowmelt or intense winter rainfall.
- Changes to hydrology that increase the probability of ground saturation.
- Land drainage schemes and river training works aimed at rapid evacuation of water, enhancing the probability of flooding and floodplain inundation within the lower catchment.

The principal drivers of enhanced coastal erosion and flooding are:

- Sea level rise (1993 to 2007 satellite altimetry indicates 3.36 ± 0.41 mm/yr (Beckley *et al.* 2007)).
- Frequency and severity of storms (North Atlantic storm wave heights have increased by 1–3 mm/yr over the last 30 years (Gulev & Hasse, 1999)).
- Expansion of coastal development and attempts at restricting coastal erosion and flooding (French 2001; McManus 2010), both of which often have unforeseen negative impacts on landforms and habitats.
- A chronic lack of sediment supply to beaches caused by near exhaustion of the sediment transported to the coastal zone during the last glacial period and exacerbated by attempts to control cliff erosion. Over the last century, the reductions in sediment supply have been manifest in coastal steepening (Taylor *et al.* 2004; Hansom 2010).

14.3.2.2 Maintenance of soil cover and low suspended sediment

The main drivers that have promoted failure in this ecosystem service are those that threaten the integrity of the vegetation cover, prolong the exposure of unprotected soil to wind and water, or lead to direct redistribution of soil (tillage erosion). In the uplands these include:

- Increased slope to channel connectivity due to drainage including moorland grips.
- Overgrazing by domesticated stock.
- Sporting endeavours, such as grouse moor burning—a substantial past driver of heather monoculture that has recently increased, along with its attendant need for burning (Ramchunder *et al.* 2009), new roads and tracks on upland peat.
- Windfarms and associated infrastructure.
- Tourism including the concentrated erosion of popular footpaths and the increased risk of fire (Cavan *et al.* 2006).
- Plantation forestry, specifically forest operations and transport network.

In grasslands, heathlands, sand dunes and machair:

- Overgrazing (including intensive winter grazing) of grasslands by domesticated stock and wild animals (Bilotta *et al.* 2007) resulting in the loss of vegetation cover and increased soil compaction. However, it is noted that 'undergrazing' in lowland heaths, resulting in undesirable habitat loss and stabilisation of soil cover in some dune systems, may reduce their protective function and habitat value.
- Intensive winter grazing.
- Frontal coastal erosion of dunes and enhanced sand blow.

In arable landscapes (Boardman & Evans 2006):

- Increase in arable area.
- Change from spring-sown to winter-sown cereals.
- Increase in row crops (maize, beet, etc.).
- High-powered equipment allowing cultivation in wetter conditions and on steeper slopes.
- Deeper, more rapid tillage on sloping ground.

- Soil compaction as a result of numerous passes of heavy machinery.
- Removal of hedges and other boundary features, and increased hydrological connectivity between fields and the fluvial system.

At a more fundamental level, the drivers of many of these changes can be found in the policy environment, especially that of UK and European Agricultural Policy. Events such as outbreaks of disease including Foot and Mouth have brought about temporary reversals of the grazing density driver.

14.3.2.3 Water retention, storage and delayed release

In addition to those drivers that threaten soil cover and low suspended sediment content, changes in water retention, storage and delayed release, evident in flooding, have been attributed to (Evans *et al.* 2004, 2008):

- an increase in impermeable areas of catchments (urbanisation, extension and increased connectivity of road networks);
- land use change that leads to reduced roughness of catchment surfaces (removal of hedgerows, overgrazing);
- isolation of rivers from their floodplains and development on floodplains;
- changes to river networks, including canalisation, that promote rapid transfer of water downstream and also increase the risk of downstream flooding and floodplain inundation;
- climate change;
- and alterations to hydrology in response to changing power demands (hydro, biofuels, etc.).

However, there are a large number of physical controls on fluvial flooding, including catchment area, topography, precipitation rates, infiltration rates, pre-existing surface and groundwater conditions, and characteristics of the river system itself. In turn, these factors are affected by climate, season, ground conditions, vegetation, land use, geology, soil structure and the existence (or not) of flood protection infrastructure, to name a few of the more important influences. The controlling processes may vary in both space and time, making floods a very complicated phenomenon to understand, capture in models and to forecast for specific scenarios.

14.3.3 Consequences of Change

14.3.3.1 Failure of maintenance of coastal land surfaces

Beaches, dunes and saltmarshes provide a sea defence role and have always been subject to evolution, but two important current changes have significant consequences. Firstly, rapid erosion, coastal flooding and landward movement of beaches, dunes and saltmarsh landforms and habitats is likely (Pye & French 1993), compromising their sea defence role. Secondly, there is now a greater concentration of human infrastructure and tangible assets at the coast and the effects of increased erosion and flooding are, therefore, more keenly felt. Assuming the present patterns continue

to be reinforced by enhanced fluvial flooding, the main erosional losers will be the landforms and habitats of the outer coast, the main gainers being landforms and habitats recreated further landward and within the inner reaches of estuaries (French 1997; Valiela 2006; Defeo *et al.* 2009). The loss of the protection provided by coastal and estuarine landforms and habitats is highly significant. Over 5 million people in England and Wales inhabit properties that are at risk of river or coastal flooding, and indicative flood maps for river and coastal flooding in the UK show large areas at risk (Environment Agency 2010; SEPA 2010a).

14.3.3.2 Failure of maintenance of sloping land surfaces and soil cover

Landslides and peatslides, although infrequent, can have catastrophic consequences as is clear from the historic record (e.g. the Aberfan disaster 1966), increased incidences of infrastructure disruption due to recent landslides in the Scottish Highlands (Winter *et al.* 2005) and the event at Dooncarton, County Mayo, Ireland, in September 2003 (Warburton *et al.* 2004); impacts of the latter included loss of all soil from large areas, loss of buried farmland, livestock, houses and a graveyard. Less dramatic, but more widespread, the loss of peatlands due to gully erosion and downwasting represents significant impacts on local biodiversity, losses of carbon storage that are unlikely to be replaceable over sub-century timescales, and the loss of sources of DOC and POC to fluvial systems. Although landslides are infrequent in the more densely populated lowlands, when muddy flows resulting from intensive rainfall on erodible soils occur, they often pass through areas of human habitation, and there have been several instances of extensive damage to civic and domestic property on large scales (Boardman *et al.* 2003). These have significant direct, indirect and intangible costs.

The continued productivity of agricultural soils does not appear to be threatened by current rates of water and wind erosion. However, the ubiquitous nature and high rates of tillage erosion may have a more deleterious impact than the more visible erosion processes (**Table 14.8**). Furthermore, the failure of the regulating services of soil retention has wider consequences; for example, fine sediment deposition in river channels, and its ingress into gravels, has been identified as a potential cause of decline in reproductive success of Salmonids. Increased sediment, nutrient and pathogen loads associated with the erosion of, and runoff from, organic and mineral soils reduce water quality and increase water treatment costs. Moreover, future changes to intensification of both plant and animal production have the potential to increase diffuse pollutant burdens.

14.3.3.3 Failure of maintenance of water retention, storage and delayed release

The Environment Agency (2009a) determined that 5.2 million homes (approximately one in six properties) in England are built on a floodplain, near a river or where there is a risk of surface water flooding during heavy rainfall. Of those properties, 490,000 are estimated to be at 'significant risk' of flooding, meaning they have a greater than one in 75 chance of being flooded in any year. In Scotland, 158,195 homes and 12,826 businesses are at risk of flooding, river flooding

alone causing around £32 million worth of damage annually (Evans *et al.* 2004; Scottish Government 2010). The costs of structural intervention (e.g. flood prevention schemes, sea walls, armoured revetments and gabions) have increased, in part because development and mitigation measures have prevented the natural landforms along rivers (floodplains and saltmarshes) and coasts (saltmarshes, wetlands, barrier beaches and dune systems) from performing their protective functions including absorbing flood waters and maintaining coastal barriers.

14.3.4 Options for Sustainable Management

14.3.4.1 Coastal protection and flood prevention

Human attempts to prevent coastal erosion and flooding have resulted in short-term success at the protection site at the expense of the long-term stability of other sites, since eroded sediments are prevented from supplying beaches, dunes and saltmarshes elsewhere and thereby sustaining their protective function. Faced with chronic negative impacts of sediment deficit and sea level rise, along with further projected changes, it is clear that the future coast cannot be accommodated within the confines of the current coastal zone and that adaptive management (managed realignment) should be pursued to allow landform and habitat to respond dynamically to the changing climate (Townend & Pethick 2002). Flooding and erosion of vulnerable, low-lying coasts puts in jeopardy the ecosystem services provided by mudflats, saltmarshes and sand dunes, so adaptive management seeks to recreate these environments landward of existing ones, allowing the coast to move in a sustainable fashion. The potential of this strategy for ecosystem services management is demonstrated by the success of the 23 coastal realignment schemes in England (Dixon *et al.* 2008) and three in Scotland. These are mostly saltmarsh recreation schemes, but there is a clear imperative to extend adaptive management to other coastal situations. Such management of ecosystem services to regulate hazard is likely to continue to be an important strategy for erosion and flood mitigation at the coast.

Similarly, in order to reduce the risk of damaging flooding within river catchments, there is a real need for adaptive management to allow river courses to move more freely and reconnect to their undeveloped floodplains. Upper catchment landforms and land uses are critical in determining the rate of transfer of water and sediment through the fluvial system to the channels and floodplains of the lower catchment. For example, there is clear evidence of the benefits of wetland management and forestry, as well as shelterbelts on agricultural land (Carroll *et al.* 2004), all increasing infiltration and reducing rapid runoff (Marshall *et al.* 2009). Such habitat recreation methods have an important role in flood risk management for the whole catchment (O'Connell *et al.* 2004; Jackson *et al.* 2006).

In relation to river flooding, the Pitt review in 2008 (an assessment of the 2007 floods) concluded that major improvements were needed at local and national level. In fact, it identified 92 recommendations that spanned all aspects of flood mitigation, adaptation, parliamentary acts,

infrastructure needs, planning, prediction and forecasting needs, emergency services, and risk assessment and prediction. Subsequent reports have addressed: national assessments of flood risk (Environment Agency 2009a, 2009b); flood and coastal risk management (Environment Agency 2009c, 2010); and guidance notes for flood preparation in Scotland (SEPA 2010b).

14.3.4.2 Erosion reduction

Sustainable management to maintain soil cover in upland areas requires an emphasis on low-intensity land use, including reducing grazing pressure and actively managing livestock. The removal of grips to re-establish wet heath and reduce the rates of water transfer to fluvial networks is expected to have significant biodiversity benefits, as well as reducing flood peak magnitude. Although the basis of this restoration measure is well-established, detailed analysis of the changes consequent to grip-blocking is required. For example, the greater potential for water storage in wet heaths may result in the quicker onset of runoff generation (due to saturation) when there is a short interval between storms, and the full effect at catchment-scale of such potential changes is uncertain. Slowing the rate of water transfer from land to fluvial systems in lowlands will be beneficial in terms of plant productivity (helping to avoid moisture shortages), processing and retention of nutrients and pathogens, reducing soil loss and reducing peak river flow (O'Connell *et al.* 2004). On grassland, the management of grazing intensity to reduce soil compaction is expected to yield benefits and the use of farm ponds and wetlands to separate and filter dirty water may reduce pathogen load and potentially enhance habitat and biodiversity (Bilotta *et al.* 2007). On arable lands, precision farming and nutrient management can reduce nutrient losses, while additions of organic matter, including on-farm products, those from civic recycling schemes and possibly biochar (Collison *et al.* 2009), can be beneficial for soil structure, reducing erosion and runoff and enhancing crop production. Contour tillage can reduce the probability of overland flow generation. Similarly, changes in tillage practice to conservation or no-till systems are likely to have benefits in terms of increasing soil quality and reducing water erosion, and would certainly reduce tillage erosion. In addition, the carbon-poor eroded elements of the landscape offer significant potential for carbon sequestration under such changed management strategies. Management of the riparian zone can alleviate some of the diffuse pollution burden—reducing nutrient, pathogen, persistent chemical and sediment delivery—and expansion of wet woods and riparian woodlands also increases habitat diversity.

14.3.5 Knowledge Gaps

14.3.5.1 Coastal erosion and flooding

It is surprising that a full assessment has not yet been undertaken of the effect that coastal change in the UK will have on erosion and flooding. For example, there remains an assumption that isostatic emergence of the Scottish coast will negate the impact of rising sea levels, whereas tide gauge data over the last 15 years shows that, in line with global estimates, all of the Scottish coast is actually subject to a 2–5 mm/yr sea

level rise (Woodworth *et al.* 2009). The availability of UK wave data is spatially patchy, as is an analysis of how enhanced wave heights, storm surges and storm frequency actually force erosion and flood-related change at the coast. Similarly, the impact of adaptive management schemes (Adger *et al.* 2005) on future tide prisms and flooding within estuaries is not yet fully understood, although there are clear habitat recreation benefits. In this context, a systematic assessment of the past and present changes to coastal landforms and habitats is crucial to better inform decision-makers about the nature of future impacts.

14.3.5.2 River flooding

Over the past 20 years there has been an explosion of risk, uncertainty and hazard-related research activity addressing flood hazards. However, hydrological risks are characterised by extreme events, for which observational data are often very limited. Even where data exist, they are rarely sufficient to characterise the behaviour of events in detail. A rigorous estimate of the possible increase in flood hazard is, therefore, a crucial task for planning future climate adaptation strategies, but our understanding of this change in risk is currently limited. Furthermore, there is limited capacity to predict propagation of flood impacts through catchments, and to mitigate them, due to the lack of multi-scale, catchment-wide monitoring and modelling (O'Connell *et al.* 2004).

14.3.5.3 Land surface stability and soil cover

In the uplands, there is a lack of systematic mapping of upland condition, although Natural England is starting to address this within its remit. Systematic monitoring could usefully address:

- GHG fluxes and carbon budgets of peatlands;
- vegetation change as an indicator of the impacts of climate change on biodiversity;
- and upland water quality, especially the concentration of phosphorus.

Although there is still much to understand about rates of ecosystem degradation, there is also a pressing need for understanding the rates and pathways of recovery from degradation:

- In relation to acidification, what are the time lags to recovery? Is there hysteresis?
- In relation to erosion-induced soil heterogeneity, how rapid is the refilling of depleted soil carbon stores and recovery of soil productivity in response to conservation measures?
- In relation to land use change for carbon sequestration (e.g. afforestation), what are the concomitant changes with respect to biodiversity, soil recovery and water quality?
- How might ongoing climate change offset or derail patterns of recovery?

There remain key uncertainties concerning residence times of soil, sediment, nutrients and pathogens passing from source to sink, and hamper understanding of the potential for their processing in temporary stores, including colluvial deposits, in-channel sediments, floodplains, wetlands and estuaries.

Neither the full biogeochemical budgets nor the optimum management of the cycles can be achieved without improved understanding of both the timescales and the processes.

14.4 Disease and Pest Regulation

14.4.1 Trends in Disease and Pest Regulation

Pathogens and pests in the UK will each be regulated or maintained below harmful levels by a specific combination of: i) biotic factors, such as predators, pathogens, competitors and hosts; ii) abiotic factors, such as climate and agricultural and urban land use; and iii) socio-economic factors, such as disease or pest management. Abiotic conditions, including fluctuating temperatures, changes in humidity or ultra-violet light (which degrades DNA), are likely to have the largest influence on pathogens that spend a large proportion of their lifecycle outside their hosts, such as vector-borne or water-borne pathogens. Biotic processes, such as genetic diversity, are particularly likely to impact multi-host pathogens and pests, especially where wild species are involved in life- or transmission cycles. Thus, the management of ecosystems and landscapes can influence the regulation of pests and diseases. As explained in Chapter 4, monitoring data are scarce for species, such as pests and pathogens, which have low cultural value. The relative role of abiotic, biotic, and socio-economic factors in regulating specific pest and pathogen systems is largely unknown, making it difficult to evaluate the importance of UK ecosystems in regulation. Here, we examine changes in incidence of pathogens and pests, and whether these changes have occurred concurrently with changes in potential drivers, in order to provide initial indications of the ecosystem components involved in their regulation and to highlight knowledge gaps.

Case studies discussed here include insect pests, and weeds of crops and pathogens of importance to humans, livestock, crops and ecosystems. Examples are restricted to those for which incidence or impacts have been altered by anthropogenic ecological changes, and which are either currently circulating in UK habitats or have a high likelihood of establishing in the medium- to long-term. An emerging disease is defined as one which has recently increased in incidence, impact, geographic or host range. Both emerging and established pests and diseases are of importance in the UK, and examples are chosen to illustrate themes in subsequent sections.

14.4.1.1 The role of biodiversity in regulating diseases and pests

Different components of biodiversity may be involved in the processes that regulate pests and pathogens in the UK: examples are presented in **Table 14.9**. Establishing these links is essential for designing intervention strategies that alter ecosystem components to enhance or reduce ecosystem processes as required.

Due to the temperate climate there are relatively few agricultural insect pests in the UK compared to continental Europe, the main group being aphids. Of the approximately 4,400 known species, around 250 feed on agricultural and horticultural crops (Blackman & Eastop 2006), to which they can cause significant damage and transmit viruses. Barley Yellow Dwarf Virus transmitted principally by *Rhopalosiphum padi* in the south of England and *Sitobion avenae* in the Midlands and north of England, causes significant crop loss in cereals (Plumb 2002). Natural enemies, such as predators, parasitoids and pathogens, are key regulators of aphids (Völkl *et al.* 2007). Exclusion of predators and parasitoids can result in reduced crop yields (Östman *et al.* 2003; Schmidt *et al.* 2003). Entomopathogenic fungi are known to cause dramatic episodes of disease in aphid populations in some years (Pell *et al.* 2001), but these are sporadic and records are infrequent.

Meta-analyses have shown that, on average, increasing the diversity of natural enemies generally strengthens pest suppression (Stiling & Cornelissen 2005; Cardinale *et al.* 2006). A greater number of predator species was found to increase aphid suppression (Snyder *et al.* 2006), and complementary effects of parasitoids and hoverflies significantly reduced aphid population growth rates (Powell *et al.* 2004). Habitat diversity at the landscape-scale is associated with an increased diversity and abundance of polyphagous predators such as beetles (MacLeod *et al.*, 2004) and especially spiders (Schmidt-Entling & Döbeli 2009). Landscapes with more field margins and perennial crops were associated with low pest establishment (Östman *et al.* 2001; **Table 14.9**). However, specific examples do not always follow this pattern; for instance, in a study of organic and conventional farms, it was observed that, overall, parasitoid diversity was greater on the organic farms, but that this did not translate into greater pest regulation (Macfadyen *et al.* 2009a). In south-west England specifically, there was no difference between organic and conventional fields in the level of cereal aphid mortality due to parasitoids, the levels of primary parasitism, hyperparasitism and multiparasitism or parasitoid diversity (Macfadyen *et al.* 2009b). Other studies have also found that species-rich parasitoid communities do not result in higher parasitism rates than species-poor communities (Rodriguez & Hawkins 2000).

Despite their importance, data on changes in the abundance of natural enemies of aphids is rare; in their study, Potts *et al.* (2006) found no significant change in natural enemies of aphids, except for Syrphidae which have been increasing since the mid-1980s, whilst MacLeod *et al.* (2004) demonstrated a maintenance of carabid diversity over seven years in a managed refuge. The status of natural enemies in providing a regulating service is, therefore, well established, but there is little information about trends over time at a landscape-scale.

One hypothesis that has been promoted is that ecosystems with greater biodiversity are more resistant to disease (Pelly 2009). Since disease outcomes result from the interactions between pathogens, vectors, hosts and the environment, the capacity of ecosystems to regulate pathogens and pests is likely to depend on several different components of biodiversity rather than simply species richness *per se*. For vector-borne diseases, one mechanism of disease reduction could be the 'dilution effect'. This is where the contact rates between vectors and competent reservoirs are reduced in a community with high host diversity either directly, due to deflection of vector meals to alternative hosts, or indirectly, because abundance of the competent reservoirs is regulated

Table 14.9 Components of biodiversity involved in regulating key pathogens and pests in the UK.

Component of biodiversity involved	Mechanism that produced the effect	Evidence from UK case studies
Habitat diversity within the landscape	Landscapes with abundant field margins and perennial crops associated with low pest establishment.	*Rhopalosiphum padi*, Östman *et al.* (2001)
	Diversity and density of natural enemies increase with perennial habitats at the landscape scale.	Spiders, Schmidt-Entling & Döbeli (2009)
	Density and species richness of natural enemies increase with provision of wildflower strips at edge of crops.	Haenke *et al.* (2009)
	Diversity and density of epigeal natural enemies increases with provision of beetle banks.	Cereal aphids, Collins *et al.* (2002)
Species richness	Greater number of predator species increases pest suppression.	Aphids, Snyder *et al.* (2006)
Host or vector population density	Increase in abundance and distribution of deer, increasing abundance of ticks (vectors).	*Lyme borreliosis*, Scharlemann *et al.* (2008), Gilbert (2009)
	Mast years increase rodent populations (reservoir species).	Hantavirus, Klempa (2009), Piechotowski *et al.* (2008)
Niche invasion or shifts in host or vector species susceptibility	Successful invasion of a sporulating host predisposing invasion of woodland and heath by pathogen.	*Phytophora ramorum* and *P. kernoviae*, Anon (2009), Webber *et al.* (2009)
		Bluetongue Virus (BTV)
Genetic diversity including human-driven genetic changes	Use of antibiotics	Antibiotic-resistant bacteria
	Vaccination	Novel strains of BTV produced by re-assortment with vaccine strains, Batten *et al.* (2008)

by competition and predation (Keesing 2006). In empirical tests, however, indices of species diversity and richness are uncorrelated (Loss 2009) or only weakly correlated with patterns in disease prevalence (LoGiudice 2008), and are less important than other components of biodiversity such as host community composition or absolute abundance of competent hosts and vectors.

Specific components of biodiversity may be responsible for the regulation of disease. An example is provided by one of the commonest tick-borne infections in the northern hemisphere, *Lyme borreliosis*: a tick-borne spirochaete infection, caused by the bacterium *Borrelia burgdorferi s.l.,* that produces viral-like meningitis and non-specific flu-like symptoms in humans. The principle vector in the UK is the sheep tick *Ixodes ricinus* which also feeds on a range of wild vertebrates. The incidence of Lyme Disease has increased dramatically over the last decade (**Table 14.10**), with the Scottish Highlands being particularly affected. Deer numbers have been positively associated with tick vector abundance as deer are key reproductive hosts for ticks (Gilbert 2009; Scharlemann *et al.* 2008).

Alternatively, specific components of biodiversity may act as a reservoir of disease, so providing 'an ecosystem disservice'. Bovine Tuberculosis (bTB) caused by *Mycobacterium bovis* is a disease which inflicts substantial economic costs on the cattle industry in the UK and could potentially affect public health. In the 1970s, infection rates had been reduced to very low levels, but they have been rising since the mid-1990s (Krebs *et al.* 1997). Human-to-human transmission is rare, with the first case being reported relatively recently (Evans *et al.* 2007). The European badger (*Meles meles*) has been implicated as an important wildlife reservoir (Woodroffe *et al.* 2006) and, in this sense, the ecosystem is providing a 'disservice'. The disease is rare or absent in many cattle-raising areas where there are no major wildlife reservoirs of disease (Krebs *et al.* 1997). In contrast, the west of England—a patchwork of agriculture, woodland, recreational countryside and residential areas—is a hotspot where protective legislation has led to an increase in badger numbers (Bourne *et al.* 2005; 2006). The distribution of the incidence of bTB in cattle has spread substantially over the last two decades, now encompassing most of the south and mid-west of England and south and east Wales.

14.4.1.2 Possible regulatory breakdown when novel pathogens invade the UK

The importance of biodiversity in regulating pathogens is perhaps best illustrated by the high prevalence and rapid spread of exotic pathogens invading new ecosystems, indicating that the usual regulatory mechanisms (predators, competitors and pathogens) have broken down. Examples include fungal plant pathogens which have changed the UK's landscape in recent years. The prime example is *Ophiostoma novo-ulmi*, or Dutch Elm Disease, responsible for killing some 30–50 million elms in the UK (Braiser 1996). Other examples include *Phytophora* species, arguably the world's most destructive group of plant pathogens; *P. ramorum* infects a broad range of plant species, including oaks, causing 'sudden oak death'. It was first detected in the UK in 2002 and has since increased in incidence in north and south-west England, although phytosanitary measures are now thought to have contained spread between nurseries, but not from nurseries to adjacent semi-natural habitats (Xu *et al.* 2009). *P. kernoviae* has more recently been detected on *Vaccinium myrtillus,* and other heathland species are known to be susceptible, leading to the suggestion that this is a potential threat for UK heathland (Beales *et al.* 2009). Since the 1990s, a stream of invasive fungal plant pathogens which are potentially damaging to trees, natural ecosystems and horticulture have been entering the UK (Brasier 2008). An agricultural example of a potential future threat is provided by *Dickeya dianthicola* (formerly *Erwinia chrysanthemi*), which causes a form of soft rot in potatoes (**Table 14.10**).

14.4.2 Drivers of Change

Over the past 60 years, land use change, specifically urban expansion and agricultural intensification (including the accidental and deliberate introduction of pest and disease organisms into natural habitats), has been the major driver of change in pest and disease incidence and, therefore, changes in their regulation. Further shifts towards the abundance of a few inimical species in the typical weed community are likely to occur while arable production is dominated by a few crops and chemical herbicides. That small changes in weed control can systematically shift the composition of weed communities, sometimes with unintended results, is shown by field experiments on Genetically Modified (GM) herbicide-tolerant winter oilseed rape in the Farm Scale Evaluations (Bohan *et al.* 2005; Squire *et al.* 2009). The herbicide used in this instance, glufosinate ammonium, encouraged grass weeds that would be detrimental to cereal crops in future years, and discouraged broadleaf weeds that support much of the arable food web. More generally, the weed flora in the UK have been resistant to the ingress of new non-crop species since seedbank records began over 100 years ago, but changes in cropping patterns and in the crop varieties grown have altered the weed flora. Crops themselves are now a common feature of seedbanks, and so, become weeds in subsequent crops. Notably, oilseed rape (**Table 14.10**) has become among the most common arable weeds since the recent rise of the crop in the 1980s (Squire *et al.* 2005). Its most prominent role might be that of an impurity in oilseed rape harvest rather than as a yield-reducing weed (Andersen *et al.* 2010).

Agricultural intensification has altered pest and disease incidence, in part because plants and animals are kept in high densities of homogenous host genotypes that favour the spread (Matson *et al.* 1997) and evolution of pathogenic strains (Slingenbergh 2004). The drive to increase the short-term profitability of milk production, for example, led to the use of ruminant-derived meat and bone-meal in animal feed and ultimately to the emergence of Bovine Spongiform Encephalopathy (BSE). The rapid spread of Foot and Mouth Disease (FMD) illustrated the large distances over which many livestock are routinely moved for trading and slaughter (Gibbens *et al.* 2001). Fish farming, if improperly managed, can increase the incidence of fish pathogens and parasites, to the extent that disease spreads to wild fish (Krkosek *et al.* 2007; Ford & Myers 2008). However, other intensive practices

Table 14.10 Status and trends of a selection of diseases of humans, livestock, honeybees and plants and weeds of crops. Values are for the UK unless otherwise specified. Honeybee sections compiled by Mike Brown, Gay Marris and Giles Budge, all of the Food and Environmental Research Agency's (Fera's) National Bee Unit.

Pathogens

Disease (Hosts)	Transmission routes	Current Incidence	Past trends	Future trends	References
Lyme Disease (Humans, deer, hare, rodents, birds)	Vector: Tick (*Ixodes* species) Broad spectrum of competent vertebrate hosts.	2007: 1.49 per 100,000, (1.34 indigenously acquired)	Five-fold increase in England and Wales, with a higher rate of increase in Scotland.	Vectors and reservoir hosts are increasing, Defra estimates up to 3,000 cases per annum, mostly undiagnosed.	Defra (2007) Kirby *et al.* (2005) Scharlemann *et al.* (2008) Health Protection Agency Milner *et al.* (2009)
Bovine Tuberculosis (bTB) (Cattle, badgers and other wildlife, humans)	Between badgers and cattle, and between cattle. Other reservoir hosts include deer.	2008: 4.9% of herds tested were confirmed as new incidents. 40,000 cattle slaughtered due to bTB in 2008.	Average annual rise of 10–11% per annum in herd incidents over the last four years. A near six-fold increase in the number of confirmed bTB reactor cattle over the 1998–2008 period.	It is thought likely that the increase in incidence in cattle will continue, probably until efficacious badger and/or cattle vaccines are developed.	www.defra.gov.uk/foodfarm/farmanimal/diseases/atoz/tb/stats/county.htm EFSA (2007)
Bluetongue (Ruminants including cattle, sheep and deer)	Vector: *Culicoides* midges	2007: 137 premises in England	First confirmed in UK in September 2007.	Vector susceptibility and population dynamics are climate mediated – therefore likely to increase in the future.	Carpenter *et al.* (2009) Purse *et al.* (2005, 2007)
Avian influenza (Wild birds, poultry)	Direct contact but can persist outside the host in water-bodies.	Prevalence in domestic birds ranges from 2% (ducks) to 50% (turkeys).	Seven outbreaks in UK poultry since 2006. One to three farms infected per outbreak.	Restoration of wetlands could potentially increase contact rates between wild and domestic birds.	Snow *et al.* (2007) Gilbert *et al.* (2008)
Chikungunya Virus	Vector: Mosquitoes with *Aedes aegyptii* being the major vector in northern Italy.	Absent in the UK	Outbreak in Italy 2007. Single mutation in the envelope protein adapted this virus to *A. albopictus*, a container breeding mosquito that has undergone a rapid spread across Europe.	Entry of Chikungunya Virus (through travel of people from endemic areas) and the vector, *A. albopictus*, into the UK in the medium term is likely. A greater frequency of warm, humid summers could facilitate establishment.	Bonilauri (2008) Knudsen (1995) Tsetsarkin *et al.* (2007) ECDC (2006) Medlock *et al.* (2006)
Gastrointestinal bacteria from farm animals (humans, farm animals)	Use of slurry on farmland, contaminated food and water, direct contact with animals, bathing in contaminated water.	*Campylobacter*: 2007: 57,590 cases Crohn's Disease: 1.6–14.7 per 100,000 people. *E. coli* O157 approx. 1,000 cases per annum. Weil's Disease (*Leptospirosis*): stable at around 50 cases a year.	*Campylobacter* has risen over the last 25 years, but has been stable over last 10 years. Incidence of Crohn's Disease is increasing worldwide (and by 4,000–8,000 people per annum in the UK), particularly in children, thought to be linked to infection with the potentially zoonotic pathogen (*Mycobacterium avium* subspecies *paratuberculosis*) which is responsible for Johne's Disease in animals. *E. coli* stable over last 10 years (rising trend in previous decade). Weil's Disease—no trend of increase in cases for the UK. Incidence decreasing in other Western	Possible increases in incidence in all cases if warm, stormy summers increase in frequency with knock-on increases in rodent populations and recreational use of fresh water but empirical associations have not been made.	EFSA (2007) Economou *et al.* (2009) Young *et al.* (2007) Rangel *et al.* (2005) Mourato *et al.* (2003) Pickup *et al.* (2006) Defra (2008a) Baranton *et al.* (2006)

	Dispersal/driver	Current incidence	Past trends	Future trends	References
Giardia *Cryptosporidium*	Contaminated food and water, direct contact with diarrhoeic human, infected pets/livestock animals. Person to person spread within families common (Caccio et al. 2005). Swimming in infected water for *Cryptosporidium*.	*Giardia*: 33,431 cases in England and Wales between 1995 and 2001 (compared with 31,655 cases of *Cryptosporidiosi*); 300–400 cases a year in Scotland (Pollock et al. 2005). *Cryptosporidium*: 3,074 cases England and Wales (2007)—average of 4,500 cases per year; 525 Scotland.	European countries over the last decades attributed to rodent control and improved hygiene. *Giardia* stable incidence since 1988 for Scotland (Pollock et al. 2005). But evidence of recent decline in cases across GB in the early 2000s (Ellam et al. 2008; Pollock et al. 2005). *Cryptosporidium* higher incidence occurring in spring and early autumn. The spring peak has been identified as predominantly *C. parvum* cases and has declined since 2001 as a result of improved drinking water quality (Lake et al. 2007).	*Giardia*: increased detection rates due to improved diagnostic tools but should also benefit from water quality regulations implemented for *Cryptosporidium*. *Cryptosporidium*: continued reduction in case numbers if drinking water quality regulations are maintained and public awareness of the risks of contact with livestock is increased.	Defra (2008a) HPA (2009) Hunter et al. (2005) Caccio et al. (2005) Lake et al. (2007) Smith et al. (2006)
Phytophthora ramorum and *P. kernoviae* (many species of shrubs and trees (>130) including *Fagus sylvatica*, *Quercus robur*, *Rhododendron ponticum*)	Splash dispersal, movement of infected material, aerial dispersal, water-borne dispersal. Spread between plants of economic and aesthetic importance aided by invasion of *Rhododendron ponticum*, a sporulating host.	*P. ramorum* has declined from 164 sites (2003) to 41 sites (2006). *P. kernoviae* first detected in heathland vegetation in 2007.	*P. ramorum* newly described in 2001. First UK record 2002. Initial spread of *P. ramorum* from garden centres probably contained by phytosanitary measures. *P. kernoviae* first described 2005.	Potential threat for UK heathland now that *Vaccinium myrtillus* infected by *P. kernoviae* has been detected in the wild, and other heathland species are susceptible.	Defra (2008b) Werres et al. (2001) Brasier et al. (2005) Beales et al. (2009) Xu et al. (2009)
Dickeya dianthicola (formerly *Erwinia chrysanthemi*) causing soft rots in potato (*Solanum tuberosum*)	Listed as a quarantine organism. Transmission via infected host, soil, water and possible alternative hosts, traced to imports of potato.	Uncertain, but probably spreading in the UK.	First reported in Netherlands in 1970s, now spreading in countries importing seed potatoes, including England and Wales; first found in English seed potatoes in 2001.	Seed certification measures as they stand may be insufficient to ensure containment; potential severe threat to UK seed potato industry, which is mainly in Scotland.	Council Directive 2000/29/EC EPPO A-2 (2009) Toth & Elphinstone (2009)

Weeds

Weed	Dispersal/driver	Current incidence	Past trends	Future trends	References
Increasing abundance and frequency of grass weeds in cereals	Seed return, dispersal; increase of cereal crops in which grasses are difficult to control.	Most arable fields	Continuing trend of increase in a range of grass species, of which *Poa annua* is now the most abundant weed in most fields.	Uncertain, depends on relative changes in phenology of weeds and crops; can be averted by varying rotations and management.	Marshall et al. (2001, 2003) Squire et al. (2005)
Herbicide resistance of grass weeds	Very high selection pressure due to use of same type of herbicide in same crop (e.g. winter wheat).	Many arable fields in the south of the UK, but recently, resistant blackgrass (*Alopecurus myosuroides*) found in Scotland.	Large increase in the UK mirrors the global trend in herbicide resistance due to large arable areas sprayed with very few herbicides (e.g. the 'glyphosate belt' in the USA).	Trend will continue unless averted by new herbicides or else more varied rotations and cropping patterns.	http://www.pesticides.gov.uk/rags. asp?id=714 ; http://www.weedscience.org/In.asp
Volunteer oilseed rape—a potential impurity in oilseed rape; a crop weed; a secondary host of *Erwinia* pathogen (blackleg in potato)	Seed drop from crops, secondary dormancy and seed persistence in soil; similar niche to annual broadleaf arable weeds; seed dropped outside farm from farm vehicles and seed transporters.	Volunteer oilseed rape in the top 10 most common arable weeds 2000–2005; established as a feral of rural landscapes across northern Europe, where it is often the commonest wayside crucifer.	Increase, from being undetected in 1970s to become commonplace as a volunteer and feral plant since the late 1980s.	Will likely be correlated with broadleaf ruderal assemblage e.g. *Capsella bursa pastoris* and *Sinapis arvensis*; long-term existence without replenishment from oilseed rape crops still uncertain.	Debeljak et al. (2008) for oilseed rape in arable seedbanks in the UK. Squire et al. (2005) for oilseed rape in the list of common weed species. Anderson et al. (2009) for examples of volunteer persistence.
Epilobium species as arable weeds	Wind dispersed, wide range of phenology, highly plastic.	Several species and hybrids now prevalent; but not yet regarded widely as a major agricultural weed.	Some types arrived in the UK about 100 years ago, but records show they were uncommon in agricultural seedbanks until recently.	Causes of spread not certain so unable to predict future trend.	Records from the Farm Scale Evaluations of GM herbicide tolerant crops.

Table 14.10 continued, Honeybees

Disease (hosts)	Transmission routes (before arrival and after arrival)	Infection route	Current UK Incidence	Past UK trends	Future UK trends	References
Varroa destructor mite (honey bees)		Direct contact (bee to bee). Adult honey bees are highly mobile hosts.	Most honeybee colonies are now classified as endemic. In some isolated areas (e.g. of Scotland) the mite may still be absent.	Confirmed in England and Wales in 1992.	Expected to remain ubiquitous but increasingly difficult to manage e.g. due to resistance and lack of effective medicines.	NAO (2009) www.nationalbeeunit.com
Tropilaelaps mite (honey bees)	Parasitic mite carried on adult bees via swarming, drifting, robbing, migratory beekeeping, and imports. Also exchange of combs and equipment and other beekeeping practices etc.	Direct contact (bee to bee). Adult honey bees are highly mobile hosts.	Apparently absent	Surveillance programme in place: Apparently absent.	Remains a high risk, two species pose risks for *Apis mellifera*. (*T. mercedesae* and *T. clareae*.) As climate warms, risks likely to increase. Formal Pest Risk Assessment in prep.	NAO (2009) www.nationalbeeunit.com Anderson & Morgan (2007)
Other parasitic mites e.g. *Varroa jacobsoni* Papua New Guinea		Direct contact (bee to bee). Adult honey bees are highly mobile hosts.	Apparently absent	Apparently absent	High risk	Anderson (2009) www.nationalbeeunit.com Pettis & Wilson (1996)
Other parasitic mites, tracheal mites		Direct contact	Widespread	Trends not monitored	Normally not high risk but synergistic effects with other pests and pathogens unknown.	
European Foulbrood (EFB) (honey bees)	Carried by adult bees via swarming, drifting, robbing, migratory beekeeping, and imports. Also exchange of combs and equipment and other beekeeping practices etc.	Larvae ingesting bacteria that are present in their food.	Approximately 4.18% of inspected apiaries in England (2008)	Trends monitored by NBU inspectorate for England and Wales. Relatively stable since 1999 (fluctuating between 2.7–4.4%).	Expected to remain stable through controls and monitoring (Statutory Notifiable disease). Large and newly detected outbreaks found in Scotland in 2009.	NAO 2009 www.nationalbeeunit.com Fera (2009)
American Foulbrood (AFB) (honey bees)		Larvae ingesting spores that are present in their food.	Approximately 0.76% of inspected apiaries (2008)	Trends monitored by NBU inspectorate for England and Wales. Relatively stable since 1999 (fluctuating between 0.1–1.2%).		

do not necessarily lead to increased disease risk; for instance, the rearing of poultry confined to indoor sheds prevents contact with wildfowl and so reduces the risk of Avian Influenza (HPAI). The increase in incidence of bTB represents another area of conflict between wildlife management (the protective legislation that has led to an increase in badger numbers) and agriculture (Bourne *et al.* 2005).

Climate change scenarios of 2°C could translate into an extra five generations a year for aphids and two to three generations for Hymenoptera—natural enemies of many pests including aphids (Yamamura & Kiritani 1998). The potential for the control of pests with natural enemies under such conditions may be variable; for example, the consumption rate of aphids by ladybirds increases more rapidly with temperature than the reproductive rate of aphids (Harrington 2002), but an asynchrony between parasitoid and host phenology could lead to a reduction in parasitism (Hassell *et al.* 1993). The extent to which this will translate into overall changes in regulatory control of aphids will depend partly on how the phenology of their crop hosts changes (including planting date), and partly on how external conditions affect the top-down control of herbivores by predators and parasitoids (Hawes *et al.* 2009). In the case of annual spring planted crops, planting dates depend greatly on soil condition in spring, which is affected by winter and spring rainfall. There is much more uncertainty over future patterns of rainfall than there is over temperature, making predictions difficult. In the case of potatoes and sugar beet in the UK, planting dates are not advancing as fast as aphid first flight dates. If this continues to be the case, aphids may arrive when crops are at an earlier and more susceptible growth stage (Harrington *et al.* 2007). Encouraging natural enemies at this time is a prerequisite for successful pest suppression by restricting early pest population increases (Landis & van der Werf 1997). Predicting the overall impact of climate change on the regulation of pests requires an integration of all these factors. A summary of possible changes in pest status due to a changing climate are presented in **Table 14.11**.

Honeybees continued						
Disease (hosts)	Transmission routes (before arrival and after arrival)	Infection route	Current UK Incidence	Past UK trends	Future UK trends	References
Predatory: Small hive beetle (honey bees)	As above but also imports of produce (fruit), imported plants (in the soil) contaminated heavy machinery and potentially in imported bumble bees.	Adults highly mobile	Apparently absent	Surveillance programme in place: Apparently absent	Remains a high risk, formal Pest Risk Assessment in prep	www. nationalbeeunit.com Cuthbertson *et al.* (2008) Neumann & Elzen (2004)
Predatory Asian hornets e.g. *Vespa velutina nigrithorax*	Imported products e.g. garden pots.	Adults highly mobile	Apparently absent	Apparently absent	Perceived to be high risk, formal Pest Risk Assessment in prep.	Chauzat & Martin (2009)
Fungal pathogens: *Nosema* species *Nosema ceranae* *Nosema apis*	Carried by adult bees via swarming, drifting, robbing, migratory beekeeping, and imports. Also exchange of combs and equipment and other beekeeping practices etc	Horizontal/Vertical (faecal/oral; genetic?)	Widespread throughout England and Wales.	Not Statutory Notifiable, trends not monitored for, but screening has shown these to be common and widespread. 35% current honey imports positive for *Nosema* species.	Unknown and under evaluation, but reported *N. ceranae* associated with losses in Spain.	www. nationalbeeunit.com Fera (2009)
Fungal pathogen Chalk brood		Direct contact; ingestion of spores.	Widespread throughout England and Wales.	Not Statutory Notifiable, trends not monitored for but screening has shown these to be common and widespread.	Impact believed to be low.	Fera (2009)
Israeli Acute Paralysis Virus (IAPV)	Carried in diseased adult bees, and by their parasitic mites, via swarming, drifting, robbing, migratory beekeeping, and imports.	Horizontal/Vertical (faecal/oral; genetic?)	Apparently absent	Not Statutory Notifiable, trends not monitored for but screening has shown apparently absent.	Remains high risk – implicated as a significant risk indicator of colony collapse disorder in the USA.	www. nationalbeeunit.com
Other honey bee viruses e.g. Deformed Wing Virus (DWV), Black Queen-Cell Virus (BQCV), Chronic Bee Paralysis Virus (CBPV)	Generally carried by adult bees and parasitic mites; therefore risk of transmission via swarming, drifting, robbing, migratory beekeeping, and imports.	Horizontal/Vertical	Widespread throughout England and Wales.	Not Statutory Notifiable, trends not monitored for, but screening has shown these to be common and widespread.	DWV correlates with poor colony health; and clear association between CBPV and colony losses in England and Wales.	www. nationalbeeunit.com

It is widely predicted that climate change will increase the incidence and intensity of diseases transmitted by arthropod vectors, such as insects and ticks (Martens & Moser 2001), but direct evidence is lacking (perhaps with the exception of Bluetongue Virus (BTV) as discussed below). Instead, there is a growing recognition that other biological and socio-economic factors may drive changes in incidence (Sumilo *et al.* 2007). Land use change is likely to have driven the increase in Lyme Disease, for example, alongside high rates of recreational use of those parts of the countryside where *Lyme borreliosis* occurs. Expansion of scrub and woodland habitat, together with milder winters and earlier springs (affecting food availability), has driven an increase in numbers of deer which are key reproductive hosts for adult ticks (Scharlemann *et al.* 2008; Gilbert 2009).

Even though climate change impacts have been detected for only a few pest and disease systems up to now, there is potential for more significant impacts in the future (Department of Health 2008). The recent succession of relatively mild winters is thought to have contributed to the increase in Lyme Disease by allowing greater tick survival. Differences in tick abundance with elevation also suggest that future climate warming could lead to further increases in tick abundance and, therefore, incidence of Lyme Disease and other tick-borne pathogens (Gilbert 2009). Evidence also indicates that the emergence of BTV—a devastating midge-borne RNA orbivirus of livestock—in southern Europe at the end of the 20th Century was climate-mediated. Non-climatic factors, such as changes in host densities and movements and agricultural land use, could not account for the spatial pattern of spread, particularly since *Culicoides* midges are generalists in their habitat and feeding preferences (Purse *et al.* 2005). Regional warming is thought to have increased the susceptibility of European midge species, widespread in the

UK, to orbivirus infection. Transmission by European midge vectors has subsequently facilitated the establishment of the disease in cooler and wetter areas in Europe (Purse *et al.* 2007). Thus, further climate change could facilitate the establishment and persistence of BTV and closely-related orbiviruses, such as African Horse Sickness Virus, in the UK.

Anthropogenic intervention has also played a significant role in suppressing pests and diseases. Typically, five to ten of the 300 or so plant species in arable seedbanks are considered noxious, economically damaging weeds (Marshall *et al.* 2001, 2003; Squire *et al.* 2005), and are usually suppressed by management (e.g. cultivation, rotations and spraying). The perennial broadleaf weed species proscribed in the various 'weed acts', such as the Weed Act 1959 and Ragwort Control Act 2003, are still sometimes injurious in grazed land, but are no longer the most damaging to

arable production, probably having been suppressed by the increased frequency of soil disturbance. The main shift in arable weed flora in recent decades has been towards grass species, probably as a result of the increasing prevalence of cereal crops that leave fewer opportunities in the cropping cycle for the control of grasses. In addition, the evolution of resistance to certain chemical herbicides has tended to increase the predominance of certain grass weeds (**Table 14.10**). During the past 25 years, the number of herbicide applications has also increased, and the active ingredients applied act upon a broader range of target species (Marshall *et al.* 2001, 2003). Consequently, most of the innocuous weed species, including those supporting the arable food web (Hawes *et al.* 2009), have declined.

Disease management has also had a substantial impact on the origin and incidence of new strains of BTV into Europe. Some live attenuated vaccines induce viraemia and

Table 14.11 Examples of insects which may increase in pest status in the UK (unless stated otherwise) under changing climatic conditions.

Pest	Potential changes	Consequence	Reference
Aphids	Increased number of generations per year, phenological shifts with earlier activity. Increased fecundity in some cases under elevated carbon dioxide.	Increased pesticide use e.g. prophylactic spraying for *S. avenae* control in autumn cereals to reduce Barley Yellow Dwarf Virus incidence. Increased risk of defoliation in spruce plantations e.g. green spruce aphid (*Elatobium abietinum*) in Sitka spruce.	Awmack *et al.* (1997) Evans *et al.* (2001) Holland & Oakley (2007) Harrington (2003) Zhou *et al.* (1995)
Lepidoptera			
Diamond back moth (*Plutella xylostella*); Silver Y moth (*Autographa gamma*)	Migrants that may be able to overwinter under warmer conditions.	Increased pesticide use resulting in increased resistance.	Cannon (1998)
Turnip moth (*Agrotis segetum*)	Increased survival of water-intolerant larval stages under drought conditions.	Increased pesticide use.	Collier *et al.* (2008)
European corn borer (*Ostrinia nubilalis*) Mediterranean corn borer (*Sesamia nonagrioides*)	Increased range into UK (in 2006 survey, restricted to South East England) from Europe as maize cropping increases.	Regular use of pesticides and/or adoption of insect resistant maize varieties.	Cannon (1998) Gianessi *et al.* (2003) Porter (1994)
Coleoptera			
Western corn rootworm (*Diabrotica virgifera*)	Increased range in UK with establishment from European base as maize cropping increases.	Increased pesticide use.	MacLeod *et al.* (2007)
Pollen beetle (*Meligethes aeneus*)	Earlier migration into crops.	Increased pesticide use and development of resistance (already seen on continent).	Holland & Oakley (2007)
Asian long-horn beetle (*Anoplophora glabripennis*) Southern pine beetle (*Dendroctonus frontalis*)	Warmer temperatures may allow establishment in Europe and may make UK forests susceptible.	Felling of diseased trees.	Ungerer *et al.* (1999) (modelling work from United States)
Pine weevil (*Hylobius abietis*)	Increased populations under warmer temperatures.	Death of re-stocked conifer plantations.	Broadmeadow & Ray (2005)
Thysanoptera			
Thrips	Warmer temperatures may allow establishment in UK.	Currently resistant to pyrethroids – increased pesticide use may increase resistance.	Defra (2006)
Diptera			
Cabbage root fly (*Delia radicum*)	Spring emergence earlier and less synchronized but total number of generations may not increase.	Increased pesticide use.	Holland & Oakley (2007) Collier *et al.* (1991)

even clinical cases of disease and can be transmitted by midges (Veronesi *et al.* 2005; Ferrari *et al.* 2005). The recent strains of BTV-6 and BTV-11 identified in the Netherlands, Belgium and Germany are closely related to such vaccine strains (Carpenter *et al.* 2009) and, since BTV is multi-segmented, may have been produced by re-assortment.

Finally, the global trade in plants and animals provides a mechanism by which pests, pathogens and vectors can enter new areas (Tatem *et al.* 2009). A lack of biosecurity, particularly in the international trade in plants, has already driven the emergence of a number of invasive pathogens in the UK, and could lead to the emergence of other diseases in the future. The potato pathogen *D. dianthicola* appears to be an example of a pathogenic organism that has been imported into the UK despite the existence of quarantine and plant health procedures. *Phytophora* species are thought to have been brought in on infected plant material, and represent one set of a number of plant pathogens which could present a threat to UK ecosystems (Brasier 2008; **Table 14.10**).

14.4.3 Consequences of Change

Any increase in pest diversity or abundance will require increased control measures; an increase in pesticide applications may be expected due to increased aphid problems, for example. As a result of a mild winter during 1988–1989, the grain aphid *S. avenae* emerged as a significant overwintering vector of Barley Yellow Dwarf Virus. In turn, this has resulted in the adoption of a prophylactic approach to autumn aphid control on cereals (Holland & Oakley 2007). Increased insecticide use may lead to increased pest resistance (Devonshire *et al.* 1998; Anstead *et al.* 2007), so integrated pest management utilising alternative strategies may become more important (Cook & Denholm 2008). A consequence of the change in the species composition of weed communities following greater use of broad spectrum herbicides is the loss of relatively innocuous weeds which play an important role supporting farmland biodiversity (Hawes *et al.* 2009).

One of the consequences of the aspiration to increase access to the countryside for recreation will be an increase in contact rates between humans and wildlife reservoirs of disease. Further trade-offs may arise with the restoration of wetlands, which could potentially augment contact zones between wild and domesticated birds. There have been seven outbreaks of the Avian Influenza Virus H5N1, and contact with wild birds has been implicated as a plausible route of entry.

The movement of plants and their products between different biogeographical zones by humans is accepted to be the primary mode by which novel pests and pathogens are introduced to new regions. If the potato pathogen *D. dianthicola* became established within the potato industry, its spread would be hard to manage and it could have a severe economic impact, particularly on the seed potato industry of Scotland. It has been argued that the current international plant health protocols are exacerbating rather than alleviating the problem, with *P. ramorum* being symptomatic of the issues. Only listed species are regulated, yet most pathogens are unknown until they have escaped their centre of origin, where they are likely to be benign, and

caused significant damage elsewhere. The consequences of leaving this particular gate open could be severe for the UK's natural environment, changing the landscape in the way that Dutch Elm Disease did in the last few decades. Change may not be fast and dramatic either—weak pathogens can contribute to declines over longer periods of time, or interact with other environmental stresses, such as climate change, (Jönsson 2004) with knock-on impacts for dependent species, amenity value and carbon storage. For example, the current death of alders due to a *Phytophora* species is damaging riparian ecosystems, destabilising riverbanks and affecting shelter for wildlife (Brasier 2008). Spread of the *Phytophthora* diseases to native heathland could have significant impacts on heathland biodiversity. The Government target of ensuring that 95% of heathland is in 'good' condition by 2010 would be jeopardised by *Phytophthora* outbreaks (Defra 2008b).

There are significant economic consequences arising from human infections acquired directly or indirectly from animals. It is difficult to estimate working days lost through gastrointestinal infections due to underreporting by patients and GP non-referrals (www.wales.nhs.uk). However, estimates from England and Wales lead to around 500,000 to 600,000 working days lost per annum (Zia *et al.* 2003; EFSA 2007). Such infections tend to be short-lived and do not need multiple drugs, apart from Crohn's Disease which is a life-long debilitating illness requiring heavy medication—the latter is estimated at costing Europe 1.6 billion Euros per year. Human health risks of bTB are relatively small due to routine milk pasteurisation and milk inspection. However, the Government spent some £108 million on control of bTB in 2008/09 (Defra 2009b), and this spend looks set to continue to increase until alternative control strategies are employed.

14.4.4 Options for Sustainable Management

Alternative agricultural management options may be required to combat any change in weed, insect pest or crop disease incidence or profile. Options are available for achieving a balance between the traits in the weed flora that reduce yield and those that support the food web. Tillage is frequently practiced to control weeds but is also responsible for loss of soil carbon, although alternatives may be sought to achieve a better balance. The arrival of novel pests may result in changes in crop varieties, for example, corn borers could be controlled by the use of Genetically Modified Insect Resistant (GMIR) maize. The limited adoption of GMIR maize in the EU is estimated to have reduced pesticide use by 53,000 kg per year (Gianessi *et al.* 2003), and evidence suggests that GMIR crops have lower impacts on natural enemies than conventional crops managed with pesticides (Marvier *et al.* 2007). Biological control as a conservation measure aims to enhance or restore indigenous populations of natural enemies through habitat management (Haenke *et al.* 2009). Although empirical evidence demonstrates the relationship between natural enemy diversity and biological control is context dependent (Straub *et al.* 2008), in general, the conservation of natural enemy biodiversity and biological control is compatible. Similarly, elements of agri-environment schemes aim to restore or preserve non-crop habitats to exploit the positive relationship that is considered

to exist between plant diversity in uncropped habitats and the diversity of natural enemies (Dennis & Fry 1992).

The development of disease risk reduction strategies, including culling of wildlife reservoirs, vaccination of susceptible hosts, reduced movement, and general management of the contact network, could form part of an organised strategy to reduce disease risk. Vaccination has been used successfully in the control of Salmonella (Mastroeni *et al.* 2001) and the 2007 BTV outbreak in England (outbreak was restricted to 137 premises (Carpenter *et al.* 2009)). But vaccination proves problematic for some other diseases of livestock, such as FMD, as it is not possible to distinguish vaccinated from recovered animals that would be routinely destroyed (Mackay *et al.* 2004). Vaccination is unlikely to be effective when wildlife reservoirs may maintain the cycle of infection (as in the case of Louping Ill and sheep, when the cycle is maintained by grouse and hares) unless these reservoirs can also be vaccinated, e.g. Lyme Disease (Tsao *et al.* 2004).

In the early part of the 20th Century, a large proportion of cattle herds in Great Britain were found to be infected with *M. bovis.* As a result, a compulsory test-and-slaughter scheme was introduced in 1950 and eventually led to the whole of GB becoming 'attested' in 1960: each cattle herd was certified as being subject to regular tuberculin testing with immediate slaughter of any reactors. A very low incidence of reactor herds was maintained throughout the 1960s and 1970s, resulting in the incidence of bTB reaching a historical low in the late 1970s and early 1980s. However, the progressive reduction in bTB incidence stalled in the mid-1980s. Since then, bTB herd incidence has remained about three times higher in parts of South West England than in the rest of GB, despite retaining an annual (and occasionally more frequent) tuberculin testing regime in those areas. The difficulties in resolving these final bTB hotspots, and the identification of a wild badger infected with *M. bovis* on a Gloucestershire farm in 1971, turned attention to the badger as a possible wildlife reservoir of infection. From 1973 to 1998, the cattle test-and-slaughter regime was complemented by a succession of culling strategies aimed at reducing badger populations in the areas where bTB remained endemic. But in the absence of experimental controls, it was not possible to know whether the observed fall in breakdowns was due to badger removal or some other factor.

Following a review by Professor John Krebs, an Independent Scientific Group (ISG) was set up in 1998 to oversee a Randomised Badger Culling Trial (RBCT) to investigate the effectiveness of badger culling (ISG 2007). As a result of work carried out in association with the RBCT it was concluded that localised culling of badgers influences their spatial organisation, disrupting territorial behaviour and increasing intraspecific transmission rates (Woodroffe *et al.* 2006). The consequence is that, while bTB incidence (confirmed cattle herd breakdowns) may be reduced within the culled area by approximately 23% (95% CI: 12.4% decrease to 32.7% decrease), it is increased in the surrounding 2 km ring by approximately 25% (95% CI: 0.6% decrease to 56.0% increase). Higher prevalence of TB among badgers may offset reduced densities achieved by culling. The end result is that badger culling has the potential to either reduce or increase

TB incidence in cattle depending on how it is carried out, its extent and its intensity (Woodroffe *et al.* 2006). The ISG's conclusion was that, to be effective, culling would have to be carried out at a landscape-scale which is unlikely to be socially acceptable, practical or economically viable (ISG 2007). There is evidence that transmission occurs from cattle to badgers, and between cattle, and that improved cattle controls might yield benefits (Jenkins *et al.* 2007); indeed, the ISG concluded that rigid application of cattle-based controls alone could prevent the rising incidence of the disease and control its geographical spread (ISG 2007). However, a consideration of this report by the then Government Chief Scientific Adviser, Sir David King, concluded that badger removal (alongside controls on cattle) remained an option for TB control if it was conducted under certain conditions, e.g. alongside hard geographical boundaries that prevent badger migration (King *et al.* 2007).

More recent analyses suggest that the negative effects of culling on disease prevalence may disappear over time (Jenkins *et al.* 2008, 2010), decreasing the landscape-scale over which the activity would need to be applied, which also changes the economic case. While badger culling remains controversial, the evidence suggests that when done on a sufficient geographical scale, in a widespread, coordinated and efficient way, and over a sustained period of time, it is likely to reduce the incidence of bTB in cattle in high incidence areas. Other control measures targeted at the transmission between badgers and cattle include the development and use of efficacious vaccines for badgers and/or cattle. An injectable badger vaccine is now licensed and is being used in a field deployment project (www.defra.gov.uk/foodfarm/farmanimal/diseases/atoz/tb/vaccination/index.htm). As a result of trade and regulatory controls, a vaccine for cattle is still considered to be some years away. It is important to note, however, that there is no single solution to tackling bTB—a comprehensive package of both cattle and badger controls, and biosecurity measures are required to effectively tackle the disease.

The movement of living plants, especially rooted nursery stock, between continents is a high risk business, and it has been argued that current biosafety protocols are insufficient (Brasier 2008). Eradication once a pathogen has arrived and established is extremely difficult and costly, so the emphasis should be on preventing pathogen introduction in the first place. Monitoring of plant material, even by molecular methods, is unlikely to be successful, due to the high quantities involved, and the lack of knowledge about which fungal isolates may be present or pathogenic. The most effective way would be to limit plant imports to licensed material: treated seeds and tissue culture only, to be propagated and tested before release (Brasier 2008). Such stringent measures would reduce, but not eliminate, the risk.

The education of health workers and GPs to recognise the most likely diseases of the future should be a priority. The Health Protection Agency estimate that most cases of Lyme Disease go undiagnosed (**Table 14.10**). Efforts to educate the public about the risk of Lyme Disease should continue, particularly in areas of relatively high incidence. Improved public awareness is also desirable for activities involving children and animals.

Horizon scanning exercises should be undertaken regularly to determine the probability of known risks from outside the UK, and combined with disease surveillance and emergency planning (HPA 2008).

14.4.5 Knowledge Gaps

The assumptions that agri-environment schemes are effective in protecting biodiversity and that enhanced biodiversity delivers enhanced pest regulation have rarely been tested (Kleijn & Sutherland 2003), although a review of Environmental Stewardship schemes suggests 87% have the potential to support biodiversity (Defra 2009b). Current empirical evidence indicates improved biocontrol of pests in some cases and negligible differences in others under Environmental Stewardship in the UK (Holland & Oakley 2007). Further data is required; natural enemy studies have predominately concentrated on parasitoids, carabid beetles and coccinellids. There is incomplete knowledge of other important natural enemies such as insect pathogens (Vega *et al.* 2009). The utility of practices such as conservation biological control could have significant positive impacts on pathogen and arthropod natural enemies (Gurr *et al.* 2004), and requires continued investigation.

The trade-offs in weed management are difficult to predict, largely because the biology of these important organisms is little known and, apart from in studies such as the Farm Scale Evaluations referred to earlier, is not prioritised by research funding bodies. For example, in the UK, the spray-area of the herbicide glyphosate rose during the 1990s at rate of about 17% per year to become the most widely applied herbicide in some regions. The impacts of this on the yield and economy of crops, weed flora and food webs, the overall profile of pesticides in water, and on the carbon footprint of arable cropping, are poorly understood, necessitating comprehensive studies on the management of weeds and other arable pests.

The identity and life history of many future pathogens are unknown to science. For example, it is estimated that only 7–10% of fungal pathogens have been identified and that 90% of pathogens are currently unknown (Hawksworth 2001). There are a large number of poorly characterised mosquito-borne viruses circulating in wild vertebrates, which could lead to emerging diseases, such as Kyansur Forest Disease, Sin Nombre Viris and Nipah, as climate and land use changes (Arinaminpathy & McLean 2009).

Recent problems with managing pathogens in the UK and Europe have arisen where the role of indigenous hosts, pathogens and vectors in the transmission cycle has not been fully explored. To forecast the likelihood of pathogen establishment, laboratory studies of host (e.g. heathland plants to *P. ramorum* and *P. kernoviae*) and vector (e.g. species within the *C. obsoletus* and *C. pulicaris* groups for BTV) species' susceptibility must be combined with detailed longitudinal field studies to understand the role of abiotic and biotic drivers in disease regulation. Certainly in the case of agricultural crop pathogens, the emphasis of research has been on epidemiology within the crop, to the neglect of biology and life history outside the crop host. This lack of fundamental knowledge makes it difficult to develop sustainable management strategies.

Obtaining information on the current distribution of infectious diseases in the UK and combining it with other spatial data on hosts, vectors and environmental conditions is currently difficult, making it hard to quantify disease changes or understand underlying processes. A previous UK Department of Health report (Kovats 2008) on the impacts of climate change on vector-borne diseases recommended the following actions:

- Create a database against which to monitor change in incidence.
- General Practitioners to report centrally on any insect-associated conditions, e.g. wasp/bee stings, rashes from caterpillars, tick bites.
- Any significant change in incidence should be the alert for focused research.
- Map the distribution and abundance of key vector species in the UK.

This should be broadened to include other pathogens with natural transmission cycles and the mapping of key domestic and wildlife hosts and other components of biodiversity involved in regulation. The European Centre for Disease Control is currently sponsoring initiatives to combine epidemiological, ecological and environmental data across Europe to understand patterns in diseases.

Another consistent feature of the case studies we have described is how landscape changes and human behavioural changes have interacted to alter the overlap between humans, hosts and vectors and the resulting contact rates between humans and pathogens. Use of key UK habitats by people, hosts, and vectors must be investigated within the same landscape framework, incorporating the work of ecologists, epidemiologists and social scientists to understand the processes underlying abrupt changes in disease incidence.

14.5 Pollination

14.5.1 Trends in Pollination

Pollination is either abiotic, primarily by wind, or biotic, primarily by bees and other insects. Pollinator-dependent crops which are restricted to enclosed agricultural land, such as oilseed rape, apples, pears and strawberries, comprised 20% of the total UK cropped area in 2007 (England: 23%, Northern Ireland: 5%, Scotland: 8%, Wales: unknown). This coverage has increased by 38% since 1989 (BHS 1999, 2008; Defra 2009c). UK consumers are highly dependent upon overseas pollination services contributing to imported foodstuffs; the UK only produces a small proportion of its own pollinator-dependent crop products (e.g. 30% of apples, 18% pears and 57% of strawberries) (Defra 2008c). Some biofuels, such as oilseed crops, require insect pollination and may become more widespread in the future if demand increases.

Using the methods of Gallai *et al.* (2009), the production function value of biotic pollination as a contribution to crop market value in 2007 was £430 million (England: £364 million, Northern Ireland: £19 million, Scotland: £47 million, Wales:

unknown), which is approximately 8% of the total value of the market (Defra 2008c, 2009b; BHS 2008; **Table 14.12**).

Wildflowers are primarily found in semi-natural grasslands, mountains, moors and heathlands, woodlands, coastal margins and some urban habitats, such as parks, gardens and roadside verges. Most wildflowers are directly dependent on insect pollination, and a high proportion of studied species (62–73%) have populations that are pollination limited (Burd 1994; Ashman *et al.* 2004). Since 1980, animal-pollinated plants have declined in the UK more than self- or wind-pollinated species (Biesmeijer *et al.* 2006), and 76% of bumblebee forage plants have decreased in frequency (Carvell *et al.* 2007).

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 & Garrod 1993; Akbar 2003; Natural England 2009).

In the UK, honeybees (*Apis mellifera*) are the most commonly managed species of pollinator, though most beekeepers are 'hobby beekeepers' primarily interested in honey production rather than providing pollination services.

Between 1985 and 2005, managed colony numbers went into serious decline (England: -54%, Northern Ireland: unknown, Scotland: -15%, Wales: -23%) and this trend is expected to continue in the short-term (Potts *et al.* 2010a; **Figure 14.3**). Feral honeybees are now almost extinct in Europe (Jaffé *et al.* 2009).

Some crops, such as strawberries, tomatoes and peppers, are mainly pollinated by managed bumblebees which are imported commercially. Though honeybees are widely managed, they are not as effective at pollinating some crops (e.g. field beans, apples, raspberry) as wild pollinators (Free 1993; Willmer *et al.* 1994; Vicens & Bosch 2000). Based on the area of crops needing insect pollination (Defra 2009b) and the recommended densities of hives needed to pollinate these crops (Free 1993), the number of registered hives in the

Table 14.12 Crop dependencies on pollinators and annual value of pollination in 2007.

Crop	Dependence on Pollinators (%)	Value per annum (£ millions)
Oilseed rape	25	106
Strawberries	45	72
Dessert apples	85	44
Culinary apples	85	43
Raspberries	45	39
Cucumbers	65	22
Tomatoes	25	21
Runner beans	85	16
Plums	65	6
Pears	65	5
Others	5–85	54
Total		**Approx. £430 million**

UK is only sufficient to supply up to a third of the pollination services, with the remainder of the services being supplied by wild pollinators. The extent to which wild pollinators contribute to wildflower pollination remains to be quantified, but is likely to be high, as species other than honeybees constitute the majority of visitors in plant-pollinator webs (Memmott 1999). Several wild pollinators are UK Biodiversity Action Plan (BAP) listed: these include 20 from a total of about 250 other species of bee, 24 from a total of 56 species of butterflies, 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).

14.5.2 Drivers of Change

Current knowledge for the UK primarily relates to the drivers of pollinators rather than the service *per se*. While global honeybee stocks have increased by approximately 45% (except in the UK where they have declined), the proportion of crops dependent upon pollination has increased much more rapidly by about 300%, meaning that demand for pollination services could outstrip the supply of honeybee hives (Aizen & Harder 2009). The drivers of the decline of managed honeybees in the UK and Europe remain poorly understood, but it is expected that a combination of pressures may be responsible (Potts *et al.* 2010a). The ectoparasitic mite (*Varroa destructor)* has contributed to the loss of most wild and feral honeybee colonies in Europe (Jaffé *et al.* 2009), and remains a major pressure on managed colonies in the UK, to the extent that it is considered impossible to eradicate (NAO 2009; **Table 14.10**). Incidences of two notifiable diseases, European Foul Brood and American Foul Brood, have remained stable since 1999 (NAO 2009; **Table 14.10**). A wide variety of other diseases are recorded for honeybees including several viruses, fungi and protozoa (**Table 14.10**). Outside the UK, no single disease can alone be attributed to recent severe colony losses (ICCL 2008), but comprehensive studies investigating colony losses across England and Wales indicate that specific organisms may be involved (www.nationalbeeunit.com); moreover, there is a trend that smaller, weaker colonies contain a greater number of disease agents. Though prevalent elsewhere, the small hive beetle (*Aethina tumida*), *Tropilaelaps* mites and Asian hornets (*Vespa velutina*) are not currently found in the UK, but remain serious threats (NAO 2009; **Table 14.10**).

Data on diseases of wild bees is lacking, but it is likely that honeybee viruses are able to invade multiple host species and thus able to infect non-*Apis* wild bees and *vice versa* (Eyer *et al.* 2009).

The loss of natural and semi-natural habitat is thought to be a major driver of wild bee declines (Winfree *et al.* 2009), but urban areas (Carré *et al.* 2009) and mass-flowering crops (Westphal *et al.* 2003) may provide important resources for some bee guilds. Semi-natural features, such as woodlands and semi-natural grasslands, have been shown to provide

a spill-over of pollinators into farmland and can increase pollination services (Kremen *et al.* 2007; Ricketts *et al.* 2008). Many important bee habitats are being lost; from 1990 to 1998, 19.8% of calcareous grasslands and 12.0% of acid grasslands have been lost from the UK (Howard *et al.* 2003). Habitat fragmentation *per se*, though not well studied, has been found to have mixed impacts on bees (Winfree *et al.* 2009). Case studies have shown that habitat degradation can be a driver of pollinator loss. Some pesticides are known to have lethal and sub-lethal effects on bees (Morandin *et al.* 2005), and intense field applications of some pesticides can cause local shifts in bee communities (Brittain *et al.* 2010). With current research to date, neonicotinoid pesticides alone appear to have no (Nguyen *et al.* 2009) or low impact on pollinators (Faucon *et al.* 2005), although no consensus across studies has been reached; most of the highly publicised cases of bee mortality are associated with spillage or incorrect use of neonicotinoids. Floral food resource availability can be reduced by herbicide use, however (Gabriel & Tscharntke 2007).

Invasive plants can have positive (Stout 2007) or negative (Traveset & Richardson 2006) impacts on native pollinators through resource supplementation and competition with native plants respectively. However, the impact of invasive plants is probably relatively minor compared to the replacement of semi-natural habitats by cereals and improved grasslands following agricultural expansion and intensification of management.

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).

Drivers of pollinator declines have been studied in isolation, but they are likely to interact in such a way that one sub-lethal driver may increase the severity of impact of another driver, and it is this that is most likely to explain the patterns of pollinator declines in the UK (Potts *et al.* 2010b).

14.5.3 Consequences of Change

Relatively little information is currently available on the direct consequences of changes in pollination services, so consequences related to changes in pollinators are reported here. 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.

Total pollinator loss for UK agriculture would translate into an annual loss of £430 million (using data from Defra 2008c, 2009b; 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. While on the one hand this is a conservative estimate of total loss value, it is an unrealistic scenario as complete pollinator collapse in the UK is improbable. Nevertheless, the calculation serves as a guide to the

relative importance of insect pollination for food production. Several studies have demonstrated that locally depauperate 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 wind- and 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. Foods particularly affected would be many fruits and vegetables (e.g. raspberries, apples, pears, cucumbers and beans) and processed and derived products.

As the majority of wild flowering plants depend upon insect pollination, decreases in pollinators will 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). Species which obligately outcross are particularly vulnerable to pollinator loss (Biesmeijer *et al.* 2006) as no compensation mechanism (e.g. self-pollination) is available. 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). Wild plants form key nodes in many food webs, and pollinator products, such as seeds and fruit, support a wide array of taxa including many invertebrates, mammals and birds. Loss of wild plants could, therefore, have wide-ranging impacts on multiple trophic levels and also negatively impact on other ecosystem services reliant on plant communities such as soil health, water quality and pest regulation. Obligately outcrossed insect-pollinated plants generally have showy flowers, and so, also enhance aesthetic values, and contribute directly to quality of life and indirectly to tourism and other rural livelihoods.

## 14.5.4 Options for Sustainable Management

Three complementary approaches can be taken to sustainably manage pollination services: a) manage the wider landscape to enhance wild pollinator populations; b) improve the health of existing managed pollinators, i.e. honeybees; and c) develop and deploy alternative managed pollinators.

a) Protection of existing, and provision of additional, high quality pollinator habitats in the wider landscape will enable more diverse and resilient pollinator communities to be supported. Key instruments include: a) pollinator-targeted options within national agri-environment schemes, such as pollen and nectar mixes for field margins (Carvell *et al.* 2007, Pywell *et al.* 2007, Potts *et al.* 2009; Chapter 6), and the Campaign for the Farmed Environment; b) cross-sectoral spatial planning to provide better landscape connectivity by allowing agri-environment scheme options to be placed strategically with respect to protected areas and other semi-natural features of the landscape; c) regulation of pesticide approval through Directive 91/414 (to be replaced by Regulation 1107/2009); and iv) continued
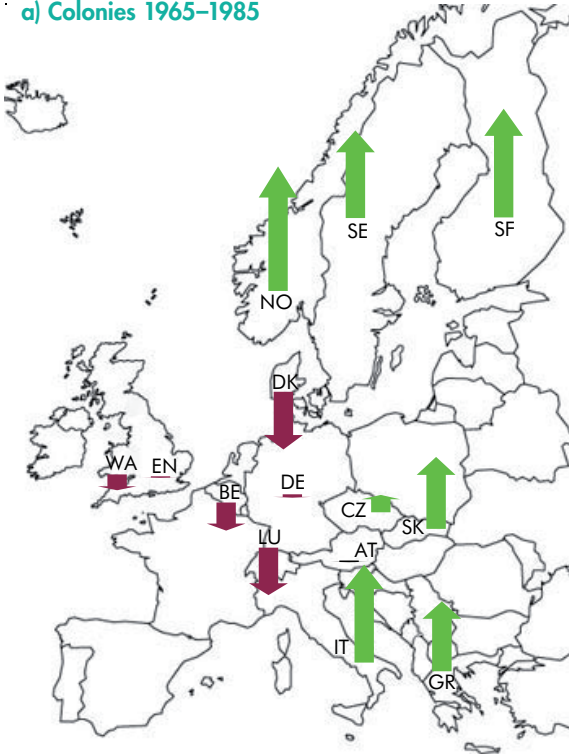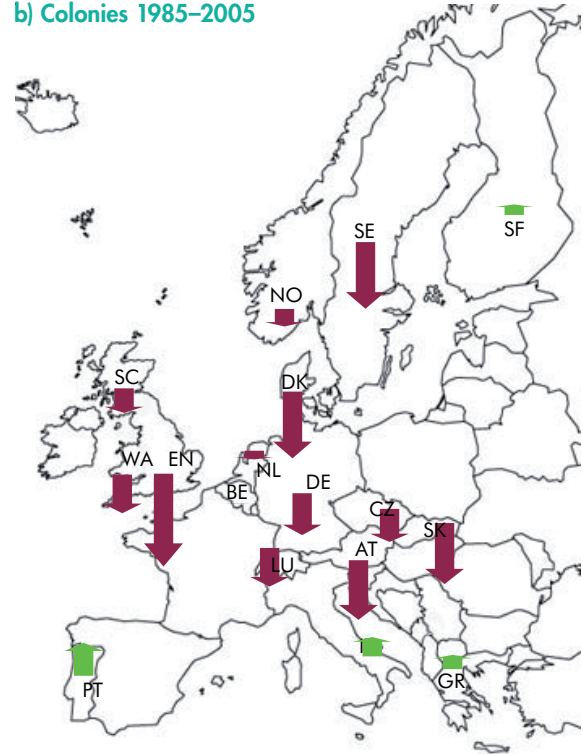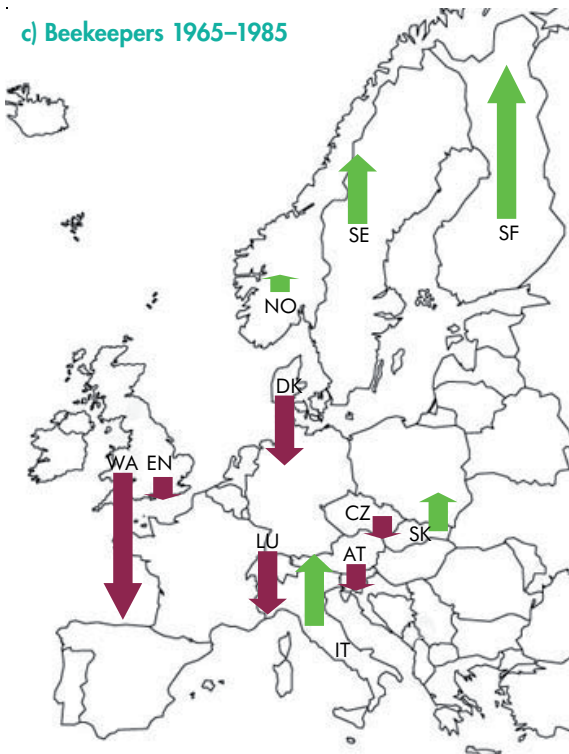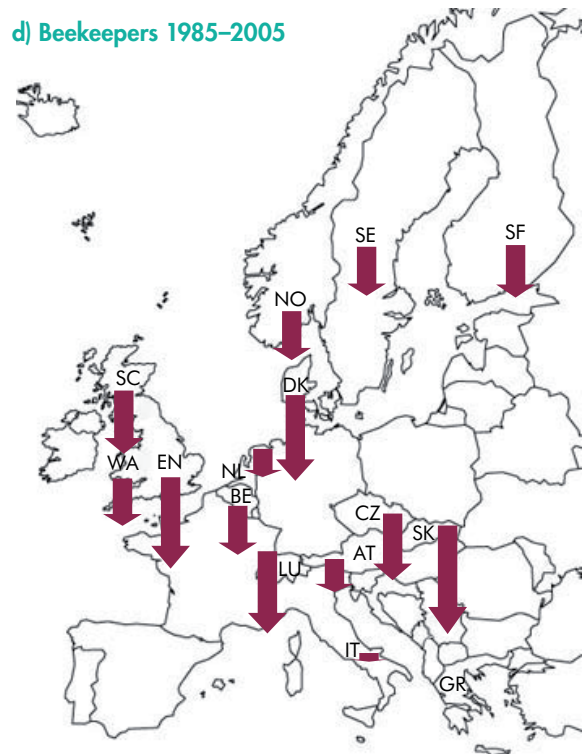
Figure 14.3 Graphical summary of the net proportional changes (%) in: a) total numbers of honey bee colonies between 1965 and 1985, and b) 1985 and 2005; c) total numbers of beekeepers between 1965 and 1985, and d) 1985 and 2005*. Maroon arrows indicate decreases, green arrows indicate increases, and the height of the arrow is proportional to the percentage change with reference arrows provided in legends. Source: reproduced from Potts *et al.* (2010a) with permission from the International Bee Research Association.

*Austria (AT); Belgium (BE); Czech Republic (CZ); Denmark (DK); England (EN); Finland (SF); Germany (DE); Greece (GR); Italy (IT); Luxembourg (LU); Netherlands (NL); Norway (NO); Portugal (PT); Scotland (SC); Slovakia (SK); Sweden (SE); Wales (WA).

support from the UK BAP, targeting actions to protect priority pollinator species and pollinator habitats. Other opportunities to support pollinators include the management of hedgerows, riverbanks, gardens and urban greenspaces to provide pollinator habitats, and the widespread adoption of pollinator-friendly practices for the application of pesticides.

b) The health of honeybee colonies can be increased through: a) improved inspection regimes including a greater proportion of colonies; b) better diagnosis of colony losses; c) a greater understanding of the drivers of honeybee loss; d) and the teaching and implementation of best practice. This requires further support for the National Bee Unit inspection programmes, and training for beekeepers. Continued stringent screening of imported queens needs maintaining to reduce the risk of small hive beetle or other new pathogens entering the UK.

c) To reduce the risks of pollination service loss we need to identify wild pollinators that can potentially be developed into managed pollinators to supplement pollination services. Reliance on a single species, honeybees, as a management solution is a high risk strategy given the increasing threats faced by this species. In addition, while honeybees are currently widely used for fruit production, pollination is often effected by forcing bees onto flowering plants by saturating the local area with hives, whereas other wild bees can be much more efficient and effective as pollinators (Willmer *et al.* 1994, Vicens & Bosch 2000). Candidate pollinators for management include several bumblebee species, the red mason bee (*Osmia rufa*) and the mining bee (*Andrena flavipes)*, the latter two having been shown to be excellent pollinators of apples and pears. Until recently, the only commercial bumblebee pollinator in the UK was a European sub-species of *B. terrestris*, which, being non-native, is only available for use in enclosed systems (e.g. glasshouses). From the spring of 2010, producers have started to supply the UK with other subspecies in limited commercial volumes for use in open-pollination systems. However, due to small production runs, these have a significant price premium. Development management of alternative pollinators requires additional funding for research in to new species, and enterprise support for species already established but requiring commercial development.

14.5.5 Knowledge Gaps

We lack national or regional monitoring schemes for both pollinators and pollination services; consequently, most of our evidence for status and trends comes from data collected for other purposes, and we have no baseline against which to compare future changes.

Basic data on the capacity of different habitats and floral resources to support pollinator communities is missing, making it difficult to model and map pollinator distributions based on land cover. Similarly, we have no generic models allowing us to predict the extent of pollinator services delivered from a given pollinator community, except in cases where a small number of well-studied species are involved.

The pollination requirements and levels of pollination limitation of many crop varieties and wild plants are poorly documented or unknown. 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 & Kremen 2009).

While many case studies have looked at single driver impacts on pollinators, the interactions of multiple drivers is poorly understood, yet this is likely to be crucial in understanding patterns of pollinator loss (Potts *et al.* 2010b). Particular needs include a better understanding of the interaction between pesticides and pathogens, and competition between managed honeybees and wild pollinators.

For honeybees, it is difficult to diagnose honeybee colony deaths with existing practices, and there are major gaps in the registration of honeybee keepers (NAO 2009); both contribute to the difficulties in identifying the causes of recent colony losses.

The combined lack of knowledge about pollinator-habitat relationships and drivers of pollinator shifts make it difficult to model future projections. Consequently, it is almost impossible to predict the likely impacts for crop and wildflower pollination, except to scope out the likely maximum impacts of total pollinator loss, which is an unrealistic scenario.

14.6 Noise Regulation

14.6.1 Trends in Noise Regulation

Sound, like light and odour, can be experienced by humans and other animals at a distance from the source. Sounds can be physically described in terms of their loudness (L), frequency and duration, but they can also have emotional and aesthetic effects (TQL 2008). Some sounds, for example bird songs, are generally perceived as pleasant and can be associated with specific ecosystems. Conversely, unwanted natural and human-derived sounds, often associated with traffic and urbanisation, are commonly termed as 'noise'. Such noise can be regulated by ecosystems by altering the sound itself or by adsorbing or reflecting the sound before it reaches the hearer.

Quantification of the status and trend of noise in the UK requires an appropriate means of measurement. Physical measurements of loudness are made on a decibel (dB) scale starting at a nominal minimum level of human hearing (0 dB). The logarithmic nature of the scale means that, for example, a sound at 60 dB is ten-times louder than a sound at 50 dB. Because human perceptions of loudness vary with sound frequency, measures of loudness are usually given an A-weighting (L_A). When the noise is discrete, such as a single aeroplane, loudness is often expressed as a maximum level (L_{Amax}). The loudness of continuous sounds, such as road traffic, may be expressed as the mean equivalent level over a period of time (L_{Aeq}).

Within the UK, major surveys of actual noise levels have been restricted to urban areas. A survey of 24-hour urban

noise levels outside 1,000 dwellings in England and Wales was conducted in 1990, and a similar study (including Scotland and Northern Ireland) was completed in 2000 (Skinner & Grimwood 2002). In both studies, daytime levels of noise were typically 8–9 dB higher than during the night (**Table 14.13**). In a comparison of sites measured in both 1990 and 2000, there was a significant decrease in daytime noise levels from 57.1 to 56.5 dB, and the level of night-time noise was stable (**Table 14.13**).

In rural areas, noise studies have been integrated with visual impact, and the studies are typically based on models where noise is related to the proximity of features such as transport corridors and population centres. The proportion of England that suffers from some degree of noise or visual disturbance has been estimated to have increased from 25% in the early 1960s, to 50% in 2007 (**Table 14.14; Figure 14.4**).

A similar trend has also been observed in Wales (WAG 2009). The effect has not been related to specific habitats.

14.6.2 Drivers of Change

Road traffic is considered to be the main source of noise in the UK (Grimwood 2002). Engine noise dominates at low speeds, whereas noise from tyres and the road surface dominates at high speeds (>50 km/h). The main driver for the modelled increase in disturbance has been the increase in population, transport and urban development. The UK's population increased from 52.8 million in 1961 to 61.4 million in 2008. The mean distance travelled per person per year increased from 3,660 miles in 1965 to 6,720 miles in 2001 (DfT 2002). These trends are expected to continue: the Department of Transport (2004) predicts a 40% increase in road traffic in England by 2025 compared to 2000 levels. The Department of Transport also predicts a two- to three-fold increase in the demand for air flights over the same period.

14.6.3 Consequences of Change

A higher proportion of people in the UK than in any other EU country identify noise pollution as one of the top five environmental issues (European Commission 2008). Noise levels can have negative effects on health, educational performance and wildlife. However, sound can also have a positive effect on human well-being.

14.6.3.1 Negative effects on health and educational performance

Exposure to noises louder than 120 dB for children and 140 dB for adults, or prolonged exposure to levels above 75 dB, can cause hearing impairment (WHO 2002). High levels of noise also reduce the intelligibility of speech, reduce cognitive performance, disrupt sleep and create hypertension. The World Health Organization (WHO 2002) also suggest that noise above 80 dB can reduce social cohesion and increase aggressive behaviour. A study in the UK has indicated that 10% of respondents 'were annoyed' when night-time noise levels exceeded 55 dB (Skinner & Grimwood 2002). Because of the increased sensitivity of humans to noise while sleeping, The World Health Organization (WHO 2002) recommends that A-weighted noise levels outside dwellings should not exceed 55 dB during the daytime and 45 dB at night. It is estimated that 54% and 67% of UK homes exceed these levels during the day- and at night respectively (**Table 14.15**).

14.6.3.2 Wildlife

While there have been various studies of the effects of noise on marine mammals and fish, the effects of noise on terrestrial biodiversity has largely focused on birds. High noise levels have been shown to decrease the food intake of some species because of the increased need for visual

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Table 14.13 Mean day-time and night-time noise levels (dB) outside of paired-samples (n = 680) of dwellings in 1990 and 2000. Source: data from Skinner & Grimwood (2002).

	1990	2000	
Day-time L_{Aeq} (07.00–23.00 hrs)	57.1	56.5	Significant (<0.05) decrease
Night-time L_{Aeq} (23.00–07.00 hrs)	48.3	48.2	No significant change

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Table 14.14 Predicted proportion (%) of each region in England disturbed by noise or visual intrusion for three selected years. Source: data from CPRE & LUC (2007).

	Early 1960s	Early 1990s	2007
North East	24.5	30.5	34.7
South West	14.6	30.1	42.5
Yorkshire & Humberside	24.0	37.1	45.9
North West	30.5	41.5	48.6
West Midlands	28.1	42.9	49.2
Eastern	21.8	38.6	49.6
East Midlands	25.8	40.9	50.2
South East & London	37.8	59.0	69.2
England	25.5	40.6	49.9

Table 14.15 Estimated proportions (%) of the population in England and Wales in 1990 and 2000, and in the UK in 2000–2001 living in dwellings exposed to levels exceeding the World Health Organization's guideline levels. Source: Skinner & Grimwood (2002).

Indicator	WHO guideline level (dB)	Proportion exceeding level (%)		
		England & Wales (1990)	England & Wales (2000)	UK (2000–2001)
Day-time L_{Aeq}, 16 hr	55	60 ± 3	55 ± 3	54 ± 3
Night-time L_{Aeq}, 8 hr	45	66 ± 3	68 ± 3	67 ± 3

scanning for predators (Quinn *et al.* 2006). In a Canadian study, industrial noise has also been related to a 17% reduction in the breeding success of a migrant bird species (Habib *et al.* 2007). Although the effect of road traffic occurs across a range of habitat types, there is particular evidence of the sensitivity of grassland birds. In the USA, Forman *et al.* (2002) related the decline in grassland birds to the increase in traffic. Likewise, Green *et al.* (2000) found that stone curlews (*Burhinus oedicnemus*) in southern England were most likely to breed in fields over 3 km away from the nearest major road. There is evidence that species vary in their response to noise, as some species can modify their signalling behaviour in terms of timing, or increased sound frequency and volume (Slabbekoorn & Ripmeester 2008).

14.6.3.3 Natural sounds and positive effects on well-being

Because most of the UK population lives in an environment of high mental stimulation, it is important that people allow for "restorative periods of cognitive quiet" or "tranquillity" (Pheasant *et al.* 2008). In the national Survey of Public Attitudes towards the Environment, tranquillity is consistently ranked as an important reason for valuing green space, along with fresh air, open space, scenery and plants and wildlife (Defra 2009d). It should be noted that tranquil spaces are not necessarily places of total silence, as tranquillity is enhanced by elements of 'soft fascination', such as a stream or a tree, which provide a pleasing level of sensory input with minimal cognitive effort (Pheasant *et al.* 2008). Similar results were derived from a national participatory appraisal which related tranquillity to both the presence of positive visual and aural factors, and the absence of negative factors (Jackson *et al.* 2008). Hearing birdsong and peace and quiet were seen as positive; hearing traffic was negative (**Table 14.16**). Using these results, and appropriate weightings for 44 indicators, relative tranquillity maps for England (500 m x 500 m resolution) have been developed (Jackson *et al.* 2008; CPRE 2009). The emphasis given to both positive and negative features means that the new tranquillity maps differ from the noise and visual intrusion maps described in **Figure 14.4.**
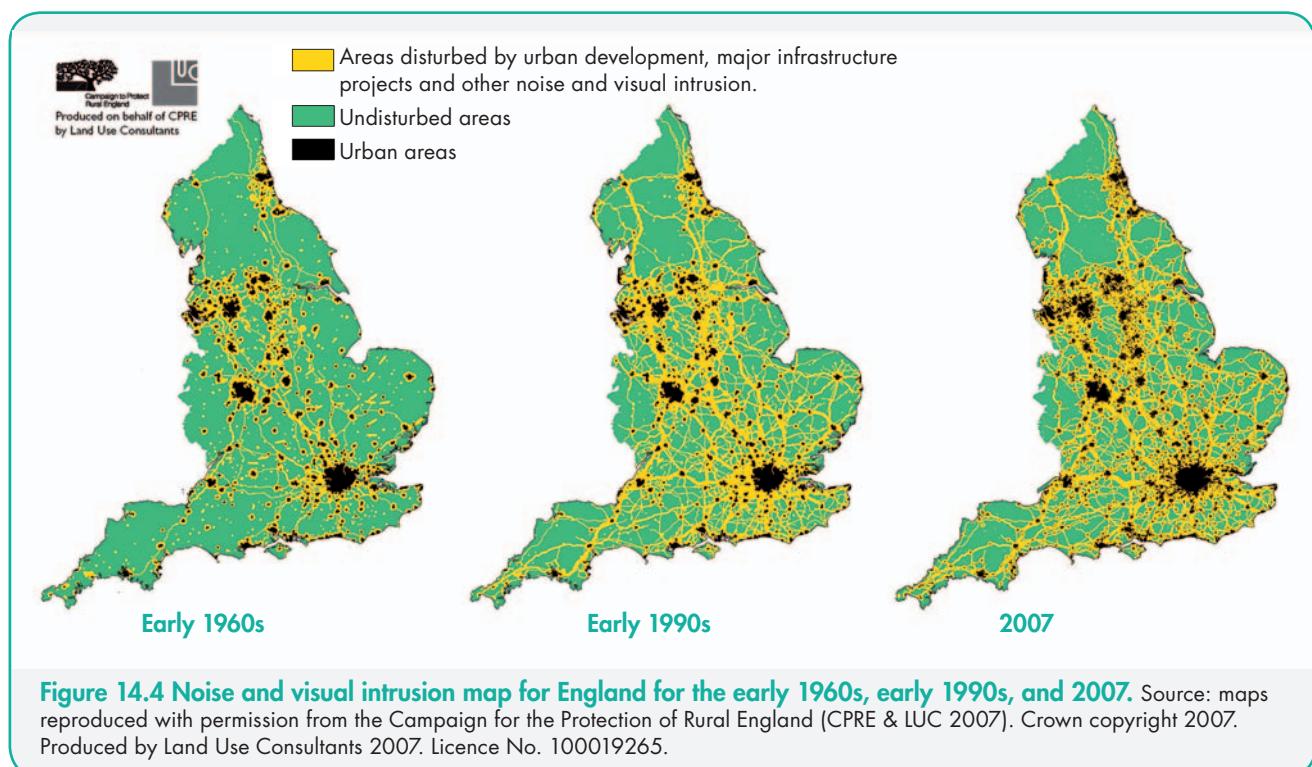
14.6.4 Options for Sustainable Management

Tranquillity is an important reason why people value green spaces and the countryside (Defra 2009d). Research has shown that people are willing to pay significant sums of money for reductions in the level of noise, and hence, noise should be a feature of the cost-benefit analysis of changes in transport infrastructure (Nellthorp *et al.* 2007). Such noise can be significant even beyond the narrow zone close to the road or railway (Countryside Agency & CPRE 2006). Methods to reduce the impact of noise in relation to traffic can focus on reducing the number of journeys, the noise per journey, or the effect of the emitted noise (**Table 14.17**). Many of these interventions act in the same way as methods to address other environmental issues such as the reduction of GHG emissions.

Table 14.16 The weighting (%) given to the top six factors (out of 44) contributing to and detracting from tranquillity as determined by 1,347 people.
Source: data from Jackson *et al.* (2008).

What is tranquillity?		What is not tranquillity?	
Seeing a natural landscape	6.6	Hearing constant noise from road transport	11.0
Hearing bird song	4.9	Seeing lots of people	7.8
Having peace and quiet	3.3	Seeing urban development	4.5

Areas disturbed by urban development, major infrastructure projects and other noise and visual intrusion.

Undisturbed areas

Urban areas

Early 1960s   Early 1990s   2007

Figure 14.4 Noise and visual intrusion map for England for the early 1960s, early 1990s, and 2007. Source: maps reproduced with permission from the Campaign for the Protection of Rural England (CPRE & LUC 2007). Crown copyright 2007. Produced by Land Use Consultants 2007. Licence No. 100019265.

Ecosystem noise regulation can be linked to changes in land form, the lateral interception of noise by vegetation, and the reduced reflection of noise by porous surfaces. Hence, the effect of noise from roads can be regulated by reducing the height of the road relative to the surrounding environment, or by constructing angled soil bunds to reflect noise upwards (Highways Agency 2008). The construction of 3–5 m-tall noise barriers, typically constructed from concrete and wood, can reduce noise levels by 8 dB(A) (Huddart 1990). Trees and other vegetation are often planted in front of such barriers to improve the visual impact. In a major review, Huddart (1990) also examined how vegetation can reduce traffic noise directly. When planted close to a road, a dense 10 m-deep row of trees was able to reduce noise levels by 8 dB(A) and 5 dB(A) compared to hard ground and grass respectively. For a 20 m-deep row, the corresponding reductions were 10 and 6 dB(A). Although requiring more space, these are similar to the reductions achieved with constructed barriers. Huddart (1990) reported that tree foliage was particularly effective at regulating high frequency sounds (>2,000 Hz), and that soft, wet, and porous surfaces were effective at reducing the reflection of low frequency sounds (<1,000 Hz). Fang and Ling (2005) reported that the greatest noise reductions occurred when the height of the tree row was at least three times that of the noise source and receiver, and when the distance between the source and receiver was less than eight times the tree height.

14.6.5 Knowledge Gaps

Spatial modelling of noise has increased substantially in response to the implementation of the 2002/49 European Directive on the Assessment and Management of Environmental Noise (END) in the UK. In 2006, maps of predicted noise levels in England were developed for 23 urban areas with populations greater than 250,000, and major roads, railways and airports (Defra 2009e). Similar maps have also been created for Scotland (Scottish Government 2007), Wales and Northern Ireland. The Directive requires the relevant authority to draw up action plans to "reduce noise where necessary" and "maintain environmental noise quality where it is good".

While spatial modelling of noise has increased, actual measurements of noise seem very limited. As shown by Skinner and Grimwood (2002), it is useful to be able to compare perceptions of noise with actual measurements. In particular, there is minimal information on the actual sound

environment in rural areas. The British Government Panel on Sustainable Development (1999) also highlights a lack of a national structure for monitoring complaints regarding traffic noise. Lastly, Payne *et al.* (2009) have highlighted the need for interdisciplinary research as the perception of noise depends on more than one sense, and there is a need to rigorously assess which interventions work and to link this with design and planning practice.

14.7 Soil Quality Regulation

14.7.1 Trends in Soil Quality

Although not explicitly mentioned in the MA, soil quality has a pivotal role in regulating services, along with air and water quality. We are reliant upon our soils to capture and release carbon, nutrients and water, detoxify pollutants, purify water, and suppress soil-dwelling pests and pathogens (Janvier 2007). The capacity of soil for regulation is determined by the interaction of its chemical composition, physical integrity and the structure and activity of soil biodiversity. Different soil types (Chapter 13) have different inherent regulating capacities. For example, certain soils are more suppressive of plant pathogens than others (Thuerig *et al.* 2009), while others provide better buffers against atmospheric pollutants (Hornung *et al.* 1995). This information is widely communicated as indicators of soil quality which are used to report on status and trends in land use, management practices and wider policy interventions. For instance, the capacity for soils to buffer against atmospheric deposition has been used to establish critical loads of atmospheric pollutants for UK habitats (**Figure 14.5**). The United Nations Economic Commission for Europe (UNECE) Convention on Long-range Transboundary Air Pollution is using information such as this to reduce emissions of air pollutants and, therefore, reduce risks to humans and the environment.

Unlike air and water quality, there is no legislation to specifically protect soil quality to maintain regulating services, although much legislation targets environmental protection *per se* and soil quality is included. The protection of soil quality is encouraged, for example, by the 2010 Soil Protection Review and associated 2010 Soil Cross Compliance Guidance, and the 2009 Defra Code for Good Agricultural Practice.

Table 14.17 Methods of reducing the emission and effects of transport noise. Source: adapted from Wright (2007) and Girvin (2009).

Means of transport	Reduce traffic flow	Reduce noise per journey	Reduce effect of noise
Road	Noise impact assessments on new roads; tolls, congestion charges; bus and high-occupancy-vehicle lane initiatives.	Speed restrictions; restrictions on engines; quieter road surfaces; tyre design.	Noise barriers; sound-proofing buildings; use of vegetation; building dwellings away from roads.
Rail	Reduced train journeys could increase road and aircraft transport.	Maintenance and repair of rolling stock.	Reduce amplification on structures.
Air	Taxes on air fuel, high occupancy initiatives.	Noise-restrictions and taxes; quiet aircraft technology; modified flight patterns.	Property acquisition around airports; timing and routing of flights; noise barriers.

Furthermore, statutory limits exist for concentrations of certain pollutants in soils to protect human health, and there are national headline indicators of soil organic matter for the Sustainable Food and Farming Strategy in England and Wales in recognition of its importance to a range of ecosystem services. Indicators of soil quality relevant to regulating services have been extensively reviewed for the purposes of UK monitoring and to aid policy decisions (Environment Agency 2006; Aalders *et al.* 2009), along with designs for a fully integrated trans-UK soil monitoring scheme. Until such a scheme is established, assessments of soil quality status and trends depend on data from large-scale and long-term surveys and experiments (Kirby *et al.* 2005; McDonald *et al.* 2007; Kirk *et al.* 2009; Emmett *et al.* 2010).

14.7.1.1 Retention, detoxification and degradation of pollutants, nutrients and carbon

Soil carbon acts as a surrogate measure for Soil Organic Matter (SOM) content which is vital for regulation. As well as binding and buffering the release of chemicals, SOM affects water retention and infiltration. Since the 1970s, the carbon content of topsoil has shown no significant change in soils under most semi-natural habitats, but small declines in arable soils (Kirby *et al.* 2005; Bellamy *et al.* 2005; Carey *et al.* 2008). There is increasing evidence that a significant proportion of arable soils are close to, or below, the critical threshold for SOM (Emmett *et al.* 2010). It remains to be determined whether exceedance of SOM levels can be linked to observed changes in soil compaction (Batey 2009), which is associated with reduced water infiltration and soil erosion from fields. In contrast to arable soils (which are generally mineral-derived) these surveys report conflicting trends in SOM and carbon for organic soils in various semi-natural habitats. These differences may reflect different sampling and analytical approaches. Given the significance of organic matter to regulating, and other, services, it is important that these issues are resolved. Forthcoming results from Scotland and the forestry sector's BioSoil Survey may help as they aim to assess trends in soil carbon data to greater depths.

Concentrations of heavy metal pollutants in soils reflect historical and current pressures alongside natural conditions. The recent UK Soil and Herbage Survey (UKSHS) (Environment Agency 2007) indicates that concentrations of copper, lead, mercury, nickel, zinc and tin are higher in urban and industrial soils than rural soils. In the main, these reflect inputs from industry and transport. Elevated levels of tin in soils of Scotland and Northern Ireland reflect local geology (Environment Agency 2007; Emmett *et al.* 2010). Recent trends in GB-wide topsoil metal concentrations from the Countryside Survey are summarised in **Table 14.18**. In general there have been relatively small changes in metal concentrations between 1998 and 2007. It is not clear what may be causing these trends—they may reflect reductions in atmospheric pollution, changes to farming practice or even reductions in SOM (as a binding agent) in arable soils. Increases within bog habitats may reflect the continued and undisturbed capture of atmospheric pollutants by organic matter.

Countryside Survey results also indicate that concentrations of heavy metals in rural soils are generally below regulatory limits, although zinc poses the most significant risk to soil organisms. Concentrations in grassland soils probably reflect sewage sludge applications, while concentrations in organic soils suggest retention of pollutants from atmospheric deposition (Spurgeon *et al.* 2008). Recent studies suggest that soil microorganisms responsible for nitrogen fixation (rhizobia) are sensitive to zinc concentrations below current statutory limits (McDonald *et al.* 2007; Defra 2009f).

The UKSHS results (Environment Agency 2007) indicate that soil concentrations of polychlorinated biphenyls in urban and industrial areas are about 1.5 times those in soils of rural areas. However, the concentrations of polychlorinated biphenyls in soil have fallen approximately 800-fold since the mid-1970s, reflecting changes to industrial production restrictions. Concentrations of total polycyclic aromatic hydrocarbons and benzo(a)pyrene in urban and industrial soils are around five to seven times those in rural areas. Regional differences in polycyclic aromatic hydrocarbon concentrations have been attributed to differences in industrial processes, fossil fuel use (e.g. coal burning and traffic) and historical agricultural inputs (Heywood *et al.* 2006). Soil dioxin levels increased between 1850 and 1980

Table 14.18 Trends from 1998 to 2007 in topsoil concentrations of heavy metals (0–15 cm) in broad habitats of GB. Arrows indicate significant increases (↑) or decreases (↓) during this period. Source: data from Emmett *et al.* (2010). Countryside Survey data owned by NERC – Centre for Ecology & Hydrology.

Broad habitat	Cadmium	Copper	Chromium	Nickel	Lead	Zinc
Broadleaved, mixed and yew woodland			↓	↓		
Coniferous woodland						
Arable and horticulture			↓	↓		↓
Improved grassland		↑	↓		↑	
Neutral grassland			↓	↓		↓
Dwarf shrub heath		↑			↑	
Fen, marsh and swamp						
Bog		↑	↑	↑		↑
All habitats		↑				

due to industrial processes, but results from the UKSHS indicate that levels have since dropped by approximately 70%. Across the UK, concentrations of dioxins in urban and industrial soils are two to three times greater than rural soils. Therefore, results indicate that UK soils are maintaining a capacity to detoxify and degrade organic pollutants. However, there is no information to indicate whether this capacity has changed in recent years, for example, through changes to soil pH and SOM.

Long-term atmospheric deposition of sulphur ('acid rain') has compromised the detoxifying and degrading capacity of UK soils with a historical lowering of soil pH throughout the country. This will have influenced the capacity to maintain other aspects of regulation such as metal retention, water purification and nutrient retention, as well as habitat integrity (**Figure 14.5**). Results from several surveys indicate that soils are now recovering from acidification, with significant increases in soil pH since the late 1970s and early 1980s

across most broad habitats (Kirby *et al.* 2005; Kirk *et al.* 2009; Emmett *et al.* 2010). These broad-scale increases reflect successful reductions in acid rain from emissions abatement (NEGTAP 2001). There are concerns that this recovery may be short-lived since continued atmospheric deposition of nitrogen will reinitiate acidification. In contrast, sulphur deficiency is being re-established in certain soils, particularly in South East England, with corresponding additional agrochemical inputs required to maintain crop production (McGrath & Zhao 1995).

Many broad habitats are now displaying nutrient enrichment through agricultural inputs and from atmospheric deposition, both point source and diffuse, which is reflected in extensive exceedance of broad habitat critical loads for nitrogen (RoTAP 2011), eutrophication issues for water quality (Section 14.7.1.2) and, more recently, changes to above-ground plant communities (Smart *et al.* 2003) and soil microbial communities (Smith *et al.* 2003).
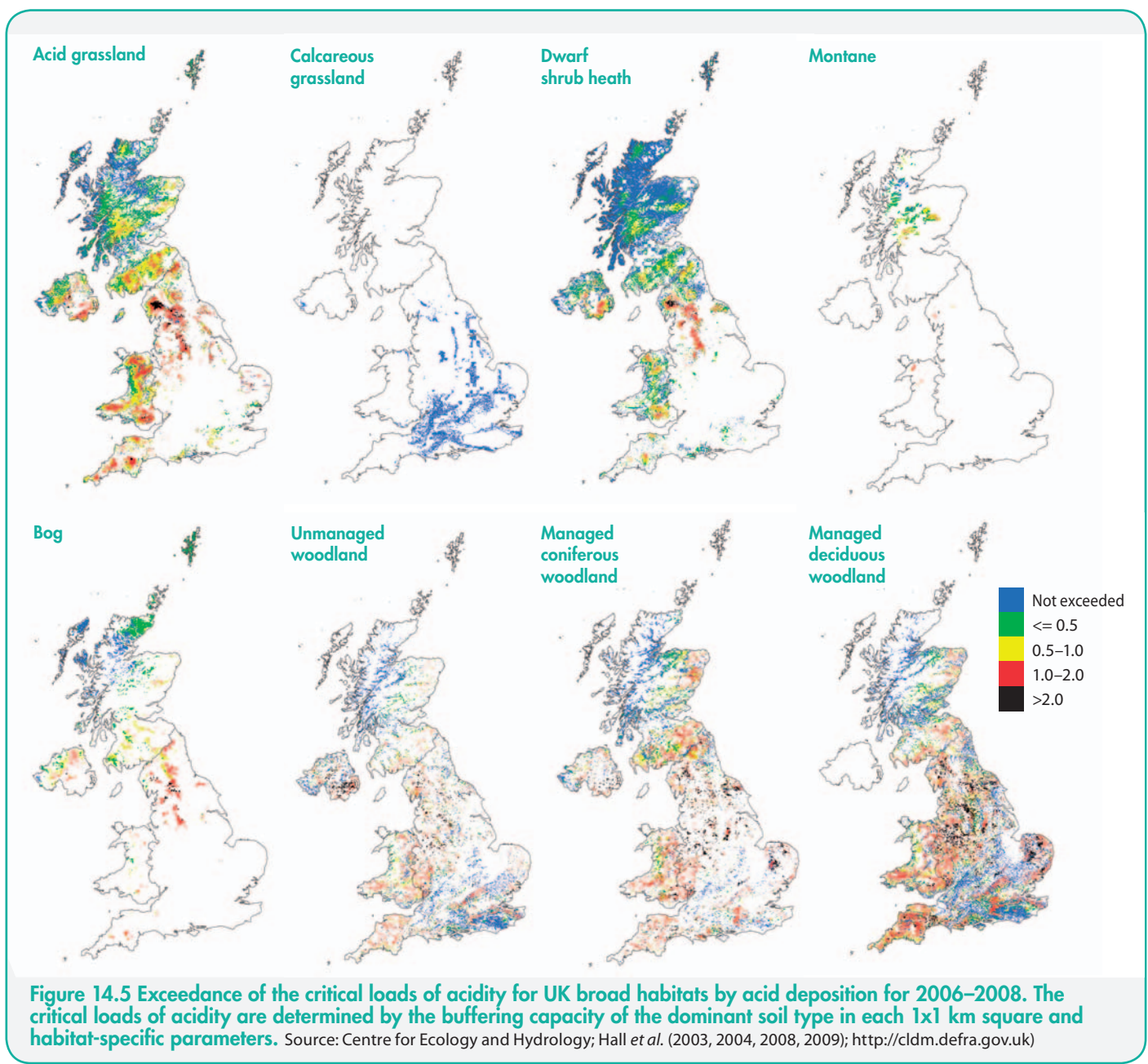

Figure 14.5 Exceedance of the critical loads of acidity for UK broad habitats by acid deposition for 2006–2008. The critical loads of acidity are determined by the buffering capacity of the dominant soil type in each 1x1 km square and habitat-specific parameters. Source: Centre for Ecology and Hydrology; Hall *et al.* (2003, 2004, 2008, 2009); http://cldm.defra.gov.uk)

These above-ground trends are not reflected in results from the Countryside Survey where topsoil nitrogen and Olsen phosphorus have both declined in almost all broad habitats from 1998 to 2007.

Historical local discharges and transboundary pollution from atom bomb-testing and nuclear incidents (e.g. Chernobyl) have resulted in elevated soil concentrations of radionuclides across the UK. Geographical patterns reflect sources as well as regional weather patterns and soil binding capacity. Soil radionuclide levels have declined in recent decades as pollutants decay and releases to the environment are reduced, with only a few locations continuing to have restricted land use from the Chernobyl fallout. These restrictions are based on a critical load approach to account for the transfer of radionuclides from soils to humans through the food chain and, more recently, for the protection of species and habitats under Natura 2000 (Environment Agency 2001).

14.7.1.2 Regulating the exchange of gases with the atmosphere

Model development and experimental studies indicate that poorly drained soils in agricultural use, in particular intensive dairy, are major sources of nitrous oxide, while organic soils with high water content are primary sources of methane (e.g. bogs and moorlands). See Section 14.2.1 for trends in GHGs in relation to land use change. There is no data on long-term trends in the capacity of soils to regulate gas exchange. Raising water tables in organic soils, as part of habitat restoration, will significantly increase methane release and should be balanced against any overall ecosystem carbon gains.

14.7.1.3 Suppressing pests and pathogens

There are distinct differences in the capacities of UK soils to suppress pathogens and pests (Section 14.5) which reflect the structure and activities of soil micro-organisms and soil chemico-physical conditions. There is no information on the current status or trends in soil suppression, which may or may not correspond to changes in the incidence of gastro-intestinal diseases involving transfers from livestock to humans via the soil.

14.7.1.4 Regulating the release of water

Regional changes to soil moisture deficits over the last 30 years reflect changing rainfall patterns (Defra 2003; Brown *et al.* 2009). However, there is a lack of information in habitat-level trends in water release from soils (Section 14.4) or in the aspects of soil quality which determine flows. Circumstantial evidence suggests that soil water retention capacity may be at risk. Results on soil bulk density from the Countryside Survey indicate that many soils in agriculture and horticulture are showing signs of reduced aeration which may influence water flow (Carey *et al.* 2008). Experimental studies indicate that reduced SOM in arable soils makes these soils more susceptible to structural damage. Evidence of localised soil compaction in agricultural soils is associated with changes in water runoff from agricultural land and in the built environment, and with increased erosion of organic soils.

14.7.2 Drivers of Change

There are numerous drivers of change in soil quality which reflect practices and objectives on many levels from local to international. For example, in direct management, farmers apply lime to alter soil buffering capacity and nutrient availability, while international regulations to control air pollution indirectly change soil quality by reducing the levels of pollutant inputs to soils from trans-boundary sources. The significance and extent of these drivers on soil quality are determined by geographical location (reflecting climate), land use, management within habitats and the resilience of individual soils to change. Soils, unlike air and water, are highly resistant to change and it can take several years or even decades for drivers of change to cause damage to soil quality, particularly where drivers are of relatively low, but repeated, intensity e.g. atmospheric inputs. There is considerable uncertainty over the timescales in, and success of, achieving recovery of soil quality for regulating services.

14.7.2.1 Agriculture and forestry

As major UK land uses, management practices within agriculture and forestry have a significant influence on all aspects of soil quality, with corresponding consequences for air and water quality (Angus *et al.* 2009; Burgess & Morris 2009). There are numerous policies, codes of practice and guidance to maintain soil quality in these land uses. Most are targeted at maintaining water or air quality or biomass production rather than soil quality *per se* (Kibblewhite *et al.* 2008). Common management practices include:

■ *Liming:* Given the acidic nature of most UK soils, liming has been a fundamental input to agricultural soils for many centuries in order to maintain an appropriate pH for biomass production and improve soil structure and suitability for metal additions. There are recognised trade-offs with liming between the maintenance of biodiversity in species-rich grassland and demands for agricultural productions. It has yet to be established whether broad-scale increases in soil pH, from reduced acid rain, is influencing the application of lime to agricultural and forest soils.

■ *Mechanisation and tillage practices:* The continuing trend for larger and heavier field machinery increases the risk of soil compaction, leading to reduced infiltration and storage of water and, potentially, increased soil erosion. Impacts of mechanisation can be minimised by use of tramlines, tracked vehicles or low pressure tyres, and corrected through cultivation and sub-soiling. Minimal cultivation systems have been proposed as an option to increase soil carbon, but there are important trade-offs to consider with soil compaction, weed control, GHG releases and yield maintenance.

■ *Drainage:* Post World War II agricultural and forestry expansion led to extensive drainage to improve soil structure and workability for crop production. It is not clear what impact this, or the subsequent removal of drainage subsidies in the 1970s, may have had on soil quality.

■ *Use of manure and artificial fertiliser*: The impacts of these on soil quality are widely documented. Technology and management tools to match fertiliser additions to soil type

and crop requirements can help minimise application rates and losses through leaching and volatilisation.

- *Simplification of cropping rotations*: A reduction in ley-arable rotations and increasing use of monocultures can influence potential pollutant levels, the suppression of pests, pathogen and weeds, and SOM levels. There is insufficient information to determine the beneficial or detrimental impacts of different cropping systems on soil quality for regulating services in the UK.

14.7.2.2 Semi-natural habitats

Drivers of change in semi-natural habitats consist mainly of the following:

- *Inappropriate burning of heather* to support game leaves soil exposed, can degrade SOM and release pollutants from the soil store to air and water.
- *Removal of historical drainage* is being used to restore regulating services in organic soils. This drainage was often installed as a precursor to land use change to forestry or agricultural intensification.
- *Restoration of vegetation.* A significant proportion of broad habitats are in poor ecological condition. Above-ground restoration activities are rather localised, but will have consequences below-ground.
- *Peat harvesting* for horticultural and fuel use. Losses of soil stock have obvious implications for soil quality
- *Recreation activities* are generally localised, but are an increasing driver in rural areas.

14.7.2.3 Universal extent

The follow drivers of change occur across all habitats:

- *Climate change, with associated policies and targets*: Although climate change has been identified as the primary threat to soils and soil quality across the UK (Towers *et al.* 2006; Defra 2009f), there are significant uncertainties over the likely responses of soils, particularly SOM and consequences for regulating services. Potential changes to the extent and management of habitats, in particular agricultural land, will have major consequences for soil quality.
- *National and EU Policy objectives:* Implementation of the Water Framework Directive and Common Agricultural Policy (CAP) cross-compliance should lead to improved soil quality for regulating services. Within England, soil policy is working to halt the decline of SOM by 2025 in arable, grassland and semi-natural habitats through improved management practices and reductions in peat-cutting. Targets for habitat restoration, particularly for the restoration of degraded bogs and heaths, will also lead to changes in soil quality. In all instances, there needs to be careful consideration of how changes to soil quality for regulating services may alter the soils' capacity to deliver other services.
- *Pollution emissions abatement:* The application of national and international regulations to limit releases of pollutants from point and diffuse sources is having considerable impacts on pollutant levels in UK soils. It is widely acknowledged that atmospheric nitrogen pollution will have to be lowered further to reduce future risks to soils, waters and habitats.

- *Soil-sealing* through rural developments, such as wind farms, and the expansion of urban and industrial areas, is recognised as a major threat to soils. For example, about 18,000 ha of previously undeveloped land were developed in England between 2001 and 2003 (Defra 2009f). Development typically removes the upper soil layers with a total loss in inherent soil quality for all services. This soil loss has been linked to increased runoff and flooding in urban areas. In recognition of this, planning and development are increasingly utilising and benefiting from the buffering and storage capacity of soils in order to deliver effective urban drainage systems. New standards have been introduced to guide the replacement of topsoil on development sites and reconstruct a more typical soil to support a range of ecosystem services.
- *Remediation of contaminated land* with targets to reduce risk from, and extent of, contaminated land will continue to improve soil quality in urban and industrial areas.
- *Rural drainage systems:* There is increasing evidence that septic tank systems from domestic dwellings are a significant source of pollutant and nutrient loadings to soils, with increasing risk of transfer to waters.
- *Overgrazing or inappropriate livestock densities* influence soil compaction, soil erosion, pathogen loadings and soil nutrient inputs. Guidance on good agricultural practice serves to limit the impact of stocking on soil quality in farmed land.

14.7.3 Consequences of Change

UK soils are already exceeding their capacity to adequately buffer ecosystems from atmospheric pollution, with well-documented consequences for air and water quality and habitat condition. This capacity has been exceeded for the majority of Broad Habitats although exceedance for individual Broad Habitats differs with geographical location; for instance, organic soils associated with bog habitats are at greater risk from atmospheric pollution in northern England than in the north of Scotland. This reduced capacity for soil buffering is linked to reductions in water quality (Section 14.7.4) and to reductions in some fungal species. There is clearly potential to improve aspects of soil quality to restore and enhance regulating services, for example, through judicious increases in SOM content and improvement in soil structural integrity. The potential benefits are clearly laid out through current estimates of the economic and environmental consequences of soil quality degradation, as illustrated below.

In the UK, the financial costs of soil carbon lost to cultivation have recently been estimated at £82 million per annum. In upland semi-natural habitats, large increases in the release of dissolved organic carbon (DOC) from organic soils since the 1980s represent a major operational cost to the water industry. The restoration of organic soils in north-west England is anticipated to save up to £2.4 million per year in avoided water treatment costs. Degradation of organic soils can also lead to losses of biodiversity, archaeological remains and paleo-climatic records, and can speed up the flow of water across the landscape, potentially increasing the risk of flooding downstream. Restoring the

soil's capacity to retain carbon will make an important contribution to climate change mitigation.

Heavy metals have the potential to irreversibly alter the soil's capacity to support ecosystems, habitats and biodiversity, and may be a risk to human health. Metal concentrations in soils are exceptionally difficult to lower once the metal has been introduced. Metals, especially zinc, affect soil respiration rates, the soil microbial community and fixation of atmospheric nitrogen. Threshold levels only exist for farmed and developed land. Ingestion by grazing animals is considered to be the pathway which holds the greatest risk of Persistent Organic Pollutants (POPs) entering the food chain. The acidification of soils can lead to detrimental effects on ecosystems, plant growth and community structure, and soil biodiversity.

The annual cost of flooding due to soil degradation is difficult to assess, with estimates ranging from £29 million to £128 million (2004/05 prices) for England and Wales alone. This does not include the impact of flooding on health, well-being and quality of life. Soil-sealing can increase rainwater runoff by as much as 50% and increases the risks of urban flooding and costs in insurance claims. Compaction decreases the infiltration capacity of the soil and increases the risk of runoff leading to flooding. It can also increase the risk of soil erosion, affect crop yields, and have a negative impact on both above-ground and below-ground biodiversity. Estimated annual costs of soil degradation are summarised in **Table 14.19**.

14.7.4 Options for Sustainable Management

UK governments and land use sectors are committed to protecting and improving soil quality for regulating and other services (Scottish Government 2009; Defra 2009g). There is a wealth of guidance on management requirements to maintain, improve or remediate soil quality for regulating services (Kibblewhite *et al.* 2008). In this context, key objectives of sustainable management practices are generally targeted at improving SOM content, improving soil structural integrity, removing or reducing pollutant loadings and toxicity levels, and retaining and utilising the soil stock. All land use sectors will increasingly need to consider how climate change will influence soil quality (e.g. soil moisture deficits, rates of soil biological processes) and management options for adaptation.

Table 14.19 Annual cost of soil degradation based on evidence presented in the Soil Strategy For England (Defra 2009f) (where values are available, but excluding contaminated land identification and remediation). Source: Defra (2009g).

Annual cost of soil degradation (£ millions)	
Soil erosion due to agriculture	45
Loss of soil carbon due to cultivation	82
Flooding due to structural damage to soil	29–128
Sediment in urban drainage systems	50–60
Total	**206–315**

14.7.4.1 Agriculture and forestry

The UK has a wealth of information and guidance for soil management in the agricultural and forestry sectors (Cranfield University 2007), and there are a multitude of management options for land managers to improve and restore soil quality. Key targets are to retain and halt the loss in SOM content and improve soil structure (Defra 2009g), with obvious benefits to regulating services. Economic drivers, alongside water quality and climate change policies, are pointing towards greater nutrient efficiencies and alternative nutrient sources which could be used to reduce nutrient and GHG transfers along with providing biodiversity benefits. Therefore, there will clearly be a need to develop alternative management strategies. Sustainable management of soil quality in the agricultural and forestry sectors now requires careful evaluation of current and alternative practices to assess the benefits and trade-offs of changing land use or practices. Lifecycle analysis and multi-criteria evaluation offer tools to explore options for balancing soil quality to maintain regulating and production services, and to ensure that management changes do not result in pollution swapping (e.g. increased GHG emissions instead of nutrient leaching) or detrimental losses to other ecosystem services.

14.7.4.2 Semi-natural habitats

In habitats of conservation interest, management strategies will be looking to reduce soil nutrient loadings to minimise impacts of eutrophication on habitats and waters. However, without further significant reductions in atmospheric nitrogen deposition, it may be impossible to achieve current conservation goals. In the short-term, therefore, site-level management will be dependent upon reliable information on current soil quality status to determine what management strategies will be feasible and achievable. Recent research has highlighted that improving the retention and storage of SOM in many habitats, in particular peatlands, will be reliant upon practices which can effectively restore soil microbial community structure (Artz *et al.* 2007). In this context, there is a need to determine how effective habitat and vegetation restoration practices (such as grip-blocking, planting, etc.) are at restoring below-ground functions, and over what timescales soil function can be restored to acceptable levels.

14.7.4.3 Urban and industrial areas

There is increasing awareness that sustainable soil use and management can benefit planning and development. Benefits to urban soil quality will be achieved through the uptake and use of the revised standards for topsoil use in developments, while retention of soil stock is recognised as an increasingly important resource for sustainable urban drainage (Defra 2009f). In parallel, there are opportunities to support expansion of allotments and urban gardens through continued practices to reduce the extent and levels of soil contamination.

Sustainable management of soils for regulating services will require consideration of the investment needed to maintain and achieve improvements in soil quality. Who bears the cost of achieving benefits if the benefits are derived

off-field or off-site? Catchment management and river basin plans for water quality improvements are going some way in dealing with this issue. Successes here demonstrate that the success of, and challenges to, sustainable management of soil quality for benefits to regulating services are the effective integration and evaluation of multiple management objectives to ensure that the outcomes are complementary and do not result in conflicting outcomes for environmental, economic and social endpoints at different spatial and temporal scales.

14.7.5 Knowledge Gaps

Compatible soil monitoring is required across the UK to provide the information necessary to assess status and trends in soil quality for regulating services. The suitability of numerous indicators of soil quality have been widely reviewed and assessed (Aalders *et al.* 2009; Defra 2009f), demonstrating recognised gaps in suitable indicators of physical soil quality (addressing soil structural changes and water transfer) and in the need for soil profile assessments. Updated information can be used to review our current interpretations of how soils are maintaining capacities to filter, buffer, transform and degrade.

Although various soil quality indicators have been identified for regulating services, data is still limited on optimal ranges, thresholds or trigger values for soil quality in individual Broad Habitats. This information will be required by land managers or other stakeholders to plan and assess the effectiveness of current or future management options to improve soil quality. The critical loads approach for atmospheric pollution has clearly demonstrated that a sound scientific understanding of the buffering capacity of soils can lead to the development of effective risk assessment and mitigation strategies at a range of spatial and temporal scales.

It is crucial that discrepancies in recent results for topsoil soil carbon across the UK are resolved. The UK is relatively rich in soils information and, with the experiences of several large-scale soil surveys, well-placed to resolve globally significant issues regarding the effective monitoring of soil carbon stocks and in improving our understanding of the sensitivity of soil carbon to climate change and other drivers.

Soil organic matter is fundamental to soil quality which underpins the role of soils in regulating services. The potential mechanisms for, and benefits of, increasing SOM in agricultural soils (and thereby reducing GHGs from soils) should be evaluated. With a significant proportion of organic soils in semi-natural habitats in a degraded state, there is a major opportunity to increase soil carbon stocks. This will require evaluation of the techniques for protecting and enhancing SOM stores over decades.

Guidance, policy and regulation will need to be evolved to provide more integrated protection to soil quality for regulating and other services, pollutants, and other pressures and drivers. This will require improvements in our capacity to predict the acceptable ranges or thresholds for pollutant loadings within individual habitats which reflect the range of services required. Site-level critical loads and the radiological Environmental Risk from Ionising Contamination (ERICA) approach are examples of how information could be used to

improve soil quality and habitat condition. In parallel, there remain key uncertainties in the mechanism transferring certain pollutants and pathogens from soils to waters.

A major challenge for protecting soil quality will be to disentangle the relative importance of direct and indirect drivers, in particular management practices, atmospheric pollution and climate change. Given the ubiquity of soils, the diversity of drivers of change and the multiple consequences of degradation in soil quality, novel approaches are now required to understand the consequences of multiple drivers across spatial and temporal scales and to keep a watching brief on emerging issues including new wastes and nanotechnologies. This will require better knowledge on the responsiveness or resilience of different soils within individual Broad Habitats in different geographical locations.

14.8 Air Quality Regulation

14.8.1 Trends in Air Quality

The main air pollutants of concern, which are considered under the national Air Quality Strategy (Defra 2007) and in international policy evaluation, are particles (PM), ozone, nitrogen oxides, ammonia and the deposition of nitrogen and sulphur. **Table 14.20** provides an overview of the major effects and trends for these air pollutants (based primarily on Defra (2007) and RoTAP (2011), and summarises the main mechanisms of ecosystem regulation.

There have been significant improvements in UK air quality over recent decades, but these have largely been driven by reductions in anthropogenic emissions such as those from transport or power generation (Defra 2007; RoTAP 2011). However, current concentrations and deposition rates still exceed thresholds for negative effects on human health and ecosystem services over significant areas of the UK. Although planned measures will further reduce emissions, substantial areas of exceedance of critical loads and critical levels (which are set as thresholds for adverse effects on sensitive ecosystems) will remain in 2020 (RoTAP 2011).

Ecosystem regulation can influence concentrations and deposition of air pollutants in four major ways. These four effects of ecosystems can be considered in terms of the impact of local anthropogenic sources, such as roads or intensive livestock units, or in terms of the transport, deposition and impacts of pollutants from a range of national and trans-boundary sources.

1. *Ecosystems remove pollutants from the atmosphere, reducing local and regional air pollutant concentrations.* Tree species are particularly effective at capturing pollutant particles and gases (Fowler *et al.* 1989; Beckett *et al.* 1998). Increasing urban tree planting and green space (Chapter 10) will, therefore, tend to increase the deposition of particulate and gaseous pollutants, and improve air quality. Recent modelling exercises provide some indication of the potential benefits. McDonald *et al.*

(2007) estimate that the current 7% tree cover in the West Midlands region reduces mean air concentrations of PM_{10} (particles above 10 μm) by 4%, and that increasing this to a theoretical maximum of 54% would reduce mean PM_{10} concentrations by 26%. The equivalent figures for Glasgow were lower, only 2% for current tree cover and 7% for maximum cover, primarily because the space available for tree planting in the city centre, where the greatest emissions and concentrations occur, is lower. The health implications of these reductions in concentrations have not been fully evaluated, although Tiwary *et al.* (2009) used a similar modelling approach to estimate the benefits of increasing urban greenspace in a 10 km grid square of East London. They calculated that two deaths and two hospital admissions would be averted each year as a result.

2. *Ecosystems contribute directly to emissions to the atmosphere.* The major contributor is the agricultural sector which is responsible for over 90% of UK emissions of ammonia. While a significant proportion of this is emitted from animal housing, mainly intensive pig and poultry units, the remainder is associated with grazing animals (primarily cattle), manure spreading and fertiliser use. These national ammonia emissions rose to a peak in 1990, but have subsequently declined, primarily because of a reduction in animal numbers and nitrogen use in fertilisers (RoTAP 2011).

3. *Emissions from ecosystems contribute indirectly to air pollution levels via chemical processing in the atmosphere.* The most important of these effects is the emission of reactive volatile organic compounds (VOCs), such as isoprene, which contribute significantly to the formation of ozone, especially during the summer (AQEG 2009). As anthropogenic emissions of VOCs fall, this source will become increasingly important. National inventories of biogenic emissions exist, but past and future trends are uncertain (AQEG 2009), although the high variation in emission rates between species means that trends in land cover are likely to influence national emissions.

4. *Measures to reduce emissions to, or deposition from, the atmosphere can increase the potential for air quality regulation by ecosystems.* This is primarily because any pollutant deposited or absorbed from the atmosphere by ecosystems is more likely to have adverse effects on those ecosystems when concentrations or deposition rates are high. An important trend in recent decades is the effect of reduced anthropogenic emissions leading to decreased urban concentrations of smoke and sulphur dioxide, which has allowed increased urban planting of conifer species. In turn, this increases interception and deposition of air pollution, hence further improving air quality. At a national scale, there is evidence that the UK landscape has become a more efficient absorber of sulphur dioxide as concentrations have fallen, further decreasing air concentrations (RoTAP 2011).

In addition to chemical pollutants, ecosystems alter air quality through the release of allergens and, in particular, pollen spores. The area affected depends on the pollen spore size; tree pollen is generally larger and travels smaller distances than grass pollen, which has a regional impact. Long-term records provide some evidence of a trend of greater and earlier tree pollen counts, but little evidence of a consistent trend in grass pollen (Spieksma *et al.* 2003).

14.8.2 Drivers of Change

The main drivers of change in air quality over recent decades have been changes in anthropogenic emissions. In terms of ecosystem regulation of air quality, two main drivers can be identified.

4.8.2.1 Land use and management

Local planting of trees and other vegetation in urban areas, especially in areas with high rates of pollutant

Table 14.20 Summary of air quality effects, trends and impacts of ecosystem regulation.

Air pollutant	Major effects	Trend since 1990	Major impacts of ecosystem regulation
Particulate Matter (PM)	Human health	Decreased concentrations	Deposition to plant surfaces, reducing air concentrations, and hence impacts on health. Emissions during fire events. Production of pollen.
Ozone	Human health. Reduced crop and forest yield. Changes in species composition.	Decrease in peak concentrations. Increase in background concentrations.	Deposition, through surface deposition and stomatal uptake, reducing air concentrations and hence impacts on health and other ecosystem services. Emission of volatile organic compounds that can contribute to ozone formation.
Nitrogen oxides Ammonia Nitrogen deposition	Human health (nitrogen oxides in urban areas). Ecosystem eutrophication Ecosystem acidification	Decrease in emissions, especially of nitrogen oxide, but only small reduction in nitrogen deposition.	Deposition, reducing air concentrations and hence impacts on health and other ecosystem services. Deposition above critical thresholds can affect sensitive aquatic and terrestrial habitats. Emissions from livestock and from use of manure and fertilisers.
Sulphur deposition	Ecosystem acidification	Decreased deposition	Deposition, reducing air concentrations and hence impacts on health and other ecosystem services. Deposition above critical thresholds can affect sensitive aquatic and terrestrial habitats.

emission, improves air quality by increasing interception and deposition (Beckett *et al.* 1998). It can also have benefits in terms of local climate and noise regulation. Tree planting around pollution sources, and more widely across the UK countryside, likewise increases removal of pollutants from the atmosphere. However, increased tree cover can have adverse effects of pollutant deposition on soils and freshwaters in sensitive catchments (RoTAP 2011) and, depending on species, may increase biogenic VOC emissions. Changes in livestock numbers, fertiliser application rates and methods of manure application will all change ammonia emission rates, and changes in land cover and grassland management can affect rates of pollen release.

4.8.2.2 Climate

Climate directly affects air quality, through changes in wind direction, atmospheric dispersal and atmospheric photochemistry. It also has indirect effects through ecosystem regulation (AQEG 2007). The most important of these effects are seen in hot dry summers. Such conditions increase the frequency of forest fires, mainly outside the UK, which can contribute to episodes of high particulate concentrations. Warmer spring weather has also led to earlier pollen release for some species, and a longer pollen season (van Vliet *et al.* 2002; Emberlin *et al.* 2002). Warmer soil temperatures can increase ammonia emissions, and also increase methane fluxes which contribute to regional ozone production. Emissions of biogenic VOCs, which lead to ozone formation, are highly temperature dependent, for example, isoprene emissions increase by a factor of four between 20–30°C (Guenther *et al.* 1993). Decreased soil water availability will close stomata and, therefore, decrease the removal of air pollutants, such as ozone and ammonia, through stomatal uptake. For instance, in the heatwave of 2003, a decrease in deposition from the atmosphere because of decreased soil water is thought to have contributed to an increase in ozone concentrations of about 20% (AQEG 2007). A further factor is the increase in atmospheric carbon dioxide concentrations. A doubling of atmospheric carbon dioxide concentrations was predicted by Sanderson *et al.* (2007) to significantly increase ozone concentrations over Europe (with adverse effects on human health) because carbon dioxide reduces the uptake of ozone through plant stomata. However, this reduced stomatal uptake by vegetation may, in turn, decrease the impact of ozone on ecosystems, including effecting crop yields and forest growth (Ashmore 2005).

14.8.3 Consequences of Change

Current levels of air pollution in the UK have significant effects on human health, crop and forest production, built materials, and biodiversity. Changes in air quality regulation can, therefore, have significant consequences for human welfare. However, the contribution of ecosystem air quality regulation to these consequences is uncertain, variable and difficult to quantify.

In terms of human health, the Air Quality Strategy (Defra 2007) estimates that the life expectancy of people in the UK is reduced, on average, by 7–8 months due to poor air quality,

although this is expected to decline to five months by 2020. These effects are mainly due to particulates; although the air quality objectives for ozone are exceeded extensively (AQEG 2009), the implications of this for human health critically depend on what threshold for effects is assumed.

Seasonal allergic rhinitis (hayfever) and other allergic diseases are directly linked to pollen levels in the atmosphere. However, while the prevalence of hayfever has apparently increased over recent decades, the levels of grass pollen have not, and hence, factors other than ecosystem regulation are likely to be involved.

In summer and winter smog episodes in particular, visibility can be decreased significantly by high concentrations of aerosols. The ecological effects of this degradation in visibility are uncertain, but the human experience of ecosystems could lose value under these conditions.

High concentrations of ozone can reduce forest growth and crop yield—losses of wheat yield in southern Britain due to ozone were estimated to be 7% in the summer of 2000, for example—and critical levels of ozone for effects on species composition are exceeded over much of the UK (RoTAP 2011).

Many sensitive lakes and streams in the UK have been significantly acidified over the last 50 to 100 years, with implications for invertebrate communities and fisheries. Chemical and biological recovery has occurred over the last two decades at non-forested sites, but evidence of recovery is more limited in forested catchments (RoTAP 2011). Although many aspects of forest management can influence acidification, afforestation is a particularly important factor as it increases interception and deposition of acidifying pollutants (Fowler *et al.* 1989).

Many terrestrial habitats are characterised by low nitrogen status. Significant reductions in total plant species diversity and the frequency of sensitive plant species have been seen in habitats with high rates of nitrogen deposition. Planned emission reductions will only reduce the area of sensitive terrestrial habitats in which critical loads for nitrogen deposition, set to prevent these effects, are exceeded from 60–49% by 2020 (RoTAP 2011). Nitrogen deposition also has wider impacts on the provision of ecosystem services (Hicks *et al.* 2008). Dry deposition of ammonia, makes a greater contribution to these adverse effects than other forms of nitrogen deposition (RoTAP 2011). Although a significant proportion of this ammonia is emitted from intensive livestock units, the remainder is mainly associated with grazing, manure and fertiliser inputs, and so, is linked to ecosystem management.

14.8.4 Options for Sustainable Management

Based on the analysis in previous sections, sustainable management for air quality regulation should maximise the removal and deposition of pollutants from the atmosphere, while ensuring that any additional deposition does not have adverse effects on other ecosystem services provided by the managed system. Management should also aim to minimise the contribution to atmospheric emissions from ecosystems. The balance between these different aims,

and the links to other regulating services, may need careful consideration. There are six priority areas for sustainable management:

1. In urban areas, tree planting can increase deposition of pollutants and improve local air quality (Beckett *et al.* 1998; McDonald *et al.* 2007; Tiwary *et al.* 2009). It also has other important benefits, such as climate and noise regulation. Ideally species should be evergreen, resistant to the pollutant loads that they intercept and efficient in intercepting particles and gases. An additional important criterion is that species should not contribute significantly to VOC emissions and, therefore, to increased ozone concentrations. Examples of species that are thought to meet these criteria include maples, birches, larches and pines, but not oaks, willows and poplars, which have high rates of VOC emission (AQEG 2009).

2. In rural areas, tree planting can also play a role in reducing the transport of pollutants from ground-level point sources, such as ammonia emissions from intensive livestock units, to sensitive local targets. Dragosits *et al.* (2006) identified that planting tree belts between large ammonia sources and sensitive sites would be beneficial in significantly reducing deposition to small, sensitive, protected sites. This planting would be more beneficial if it was located around the protected site rather than around the source.

3. Wider-scale rural tree planting will also increase the deposition of pollutants from the atmosphere, reducing air pollutant concentrations. However, in regions with sensitive catchments, it is important that the deposition of sulphur and nitrogen from the atmosphere to soils (and through leaching into freshwaters), does not exceed critical loads. Similarly, future tree planting at a UK scale needs to carefully consider the implications for VOC emissions as several species considered for short-rotation forestry (such as poplars and willows), are high emitters of VOCs which contribute significantly to ozone production.

4. In arable, improved and semi-natural grassland habitats, sustainable management is needed to reduce ammonia emissions to the atmosphere without leading to adverse effects on other ecosystem services, for example, increasing nitrate leaching into waters. This might involve management measures such as the rapid incorporation of manure and slurry applications into arable soils, replacement of urea fertilisers with ammonium nitrate, and changes in cattle diets to reduce nitrogen excretion (Misselbrook *et al.* 2009). Management at a local level can also be used to protect sensitive sites; Dragosits *et al.* (2006) identified the value of local buffer zones of low-emission agriculture around sensitive sites in reducing ammonia deposition.

5. Sustainable management measures that increase water conservation and reduce fire risks will have benefits in terms of air quality regulation, especially in minimising the negative impacts of warmer, drier summers. Such climatic conditions will lead to a decreased deposition to land surfaces and higher atmospheric concentrations of pollutants such as ozone, as well as an increased likelihood of fires, leading to increased particle concentrations (AQEG 2007).

6. Grassland management, in particular the timing of cuts and the intensity of grazing, and the choice of seed mix, can play a significant role in reducing levels of grass pollen in the atmosphere (Emberlin *et al.* 1999).

14.8.5 Knowledge Gaps

In general terms, we understand the basic mechanisms whereby ecosystems influence and regulate air quality. Modelling tools have been developed to quantify these effects at a local or urban level for individual pollutants, and their response to planned or unplanned interventions. However, we lack the modelling tools to provide an integrated assessment for all the major pollutants at a national scale, and hence, an overall assessment of the value of air quality regulation. Furthermore, methods of linking the impacts of ecosystem air quality regulation on local and regional air quality to the impacts of air quality on human health, crop and forest production, carbon sequestration and ecosystem integrity need to be developed for a full evaluation to be made.

Specific knowledge gaps that need to be addressed in addition to these broad objectives include:

- *The regulation of different particle sizes:* The deposition velocity of particulate matter to vegetation is strongly dependent on the particle size distribution, with deposition of very coarse and ultra-fine particles being greatest. While air quality standards to protect human health are set on the basis of concentrations of PM_{10}, there is increasing evidence that ultra-fine particles, which can penetrate deeper into the lung and may cross into the blood stream, are more closely related to health effects. Deposition to vegetation is more efficient for fine particles (smaller than 1 μm) than for particles with a diameter in the size range 1–10 μm (Freer-Smith *et al.* 2005). Hence, the benefits of tree planting in urban areas may have been underestimated by studies to date which have focused only on PM_{10}. Therefore, more detailed evaluation is needed of the local and national benefits of ecosystem regulation for different particle sizes.

- *Knowledge of natural emissions of VOC*: Knowledge in this area is not adequate for the modelling and assessment of ground-level ozone as estimates are only available of annual totals (Royal Society 2008). Improved, spatially and temporally disaggregated, emission inventories and parameterisations for these compounds from UK vegetation need to be developed (RoTAP 2011).

- *Better description of the deposition of ozone:* The stomatal and non-stomatal deposition of ozone to terrestrial surfaces exerts a strong control on ground-level concentrations, especially in peak ozone episodes. Better description of these deposition processes, and their effects on concentrations, are needed for air quality assessment models (RoTAP 2011). In particular, the significance and control of non-stomatal deposition of ozone, and whether it is due to thermal decomposition (Fowler *et al.* 2001) or reactions with biogenic VOC emissions (Goldstein *et al.* 2004), needs better understanding.

14.9 Water Quality Regulation

14.9.1 Trends in Water Quality

Water quality is determined primarily by catchment processes including plant and microbial nutrient uptake, pollutant accumulation in SOM and adsorption on to mineral surfaces, acidity buffering, organic pollutant breakdown, and denitrification. If the capacity of plants and micro-organisms to assimilate, buffer or process pollutants, or of soils to sequester them, is exceeded, water quality is degraded. Drainage systems themselves can dilute pollutants (e.g. from point sources) to safe levels; process or assimilate them (e.g. in lakes and wetlands); or simply transfer pollutants from one ecosystem (the land) to another (the ocean).

There is little, if any, direct long-term monitoring of the ecosystem processes which regulate water quality (Chapter 9). Water quality monitoring data have, therefore, been used to provide inferences regarding these processes. Most monitoring by national environmental agencies is targeted at rivers draining larger, mixed catchments, which are strongly influenced by direct pollutant inputs. Chemical General Quality Assessments, which measure organic pollution, showed that 79% of English, 95% of Welsh and 74% of Northern Irish rivers, by length, were of good chemical quality in 2008, compared to only 55%, 86% and 44%, respectively, in 1990 (Defra 2009h). Although Scottish data cannot be compared directly, a combined chemical, biological and aesthetic assessment classified 88% of rivers as good condition in 2008. Concentrations of phosphates were high (>0.1 mg phosphorus/litre) in 52% of rivers, by length, in England, 23% in Northern Ireland, 4% in Scotland, and 8% in Wales. Nitrates, which cause eutrophication in coastal waters and can impair drinking water quality, showed a similar pattern, with high concentrations (>30 mg nitrate/l) in 32% of English rivers, by length, compared to ≤3% elsewhere in the UK.

The Harmonized Monitoring Scheme (HMS), comprising 230 large river sites, provides the best available long-term perspective on water quality trends in large British rivers. Organic pollution indicators (biological oxygen demand and ammoniacal nitrogen) have declined everywhere since the 1980s, leading to increased dissolved oxygen concentrations (**Figure 14.6**). Phosphate concentrations have fallen substantially in most areas since the mid-1990s, while nitrate has shown little change. Concentrations of heavy metals have generally declined. Monitored pesticides are below detection limits in most samples and clear trends cannot be discerned, but three pesticides (isoproturon, mecoprop and diuron) were above 0.1 µg/l (an arbitrary threshold) in >10% of Environment Agency samples from 1995 to 2004. Faecal indicator organisms (FIOs) are measured routinely at bathing waters which are predominantly coastal. Coastal waters complying with mandatory standards set by the EC Bathing Water Directive increased from 74% in the period 1988 to 1991 to 98% in 2004 to 2007. The stricter Guideline Standards, which indicate excellent bathing water quality, were met in 37% of waters in 1994 to 1997, rising to 73% in 2004 to 2007.

Water quality is more strongly linked to ecosystem processes in the uplands than in the lowlands, but upland waters are poorly represented by standard agency monitoring regimes. While less impacted by agriculture, upland waters are subject to other environmental pressures and are important aquatic habitats (they include many of the UK's lakes) and a major source of clean drinking water. Being relatively unpolluted, water draining the uplands performs a key regulatory service by diluting pollution that enters river systems further downstream. Many upland waters were damaged by acidification during the 1960s to 1980s, but partial recovery is now evident, with pH increasing and toxic aluminium concentrations declining (AWMN 2009; **Figure 14.7a**). Waters draining peaty soils are high in dissolved organic carbon (DOC), and concentrations have risen across the UK in the last 20 to 30 years (Evans *et al.* 2005a; Worrall & Burt 2007; **Figure 14.7b**). Suspended sediment levels are elevated in areas of active soil erosion, notably downstream of eroding peatlands such as those of the South Pennines (Evans *et al.* 2006). Upland waters are typically low in nutrients, although nitrate concentrations are higher in areas receiving elevated atmospheric deposition (Allott *et al.* 1995). Since the 1980s, there have been no clear trends in upland nitrate concentrations (Curtis *et al.* 2005).

14.9.2 Drivers of Change

In lowland catchments, by far the dominant driver of water quality change over the last 30 years has been regulation of pollutant inputs, rather than changes in ecosystem regulating processes. The greatest improvements are associated with regulation of large point sources, such as waste water treatment works (WWTWs), whilst small point source and diffuse inputs remain comparatively high. Point sources of phosphorus are still considered to provide the greatest risk of eutrophication in rivers, even in rural areas (Jarvie *et al.* 2006; Arnscheidt *et al.* 2007). Since 1984, phosphorus fertiliser applications have fallen by 53%, and nitrogen applications by 32% (Defra 2009h). Nitrate concentrations have not decreased, however (**Figure 14.6**), suggesting that the capacity of agricultural ecosystems to retain excess nitrogen has, if anything, declined over the period.

Discharge rates are key to water quality. Low flows reduce effluent dilution capacity and can lead to ten-fold increases in nutrient and organic pollutant concentrations. During summer droughts, 25–33% of flow in some rivers in central and south-east England can be comprised of sewage effluent. High flows mobilise and transport bacterial pollutants from agricultural sources and combined sewage overflows, as well as suspended sediments and associated pollutants. Discharge is affected by climate, particularly the frequency and amount of precipitation, abstraction and land management. Rising water temperatures may accelerate biological activity and nutrient cycling, and exacerbate poor water quality at low flows by reducing dissolved oxygen. Future climate change projections suggest that the frequency of algal blooms will increase, with 70–100% higher pollutant concentrations (Johnson *et al.* 2009).

The dominant driver of changes in upland water quality in the past has been atmospheric deposition. Sulphur emissions from fossil fuel burning, the primary agent of

acidification, have decreased by 90% since 1970, and should fall further by 2020 (RoTAP 2011). In many areas, acid loadings are now less than the buffering capacity provided by base cation weathering, leading to freshwater recovery. However, the historic depletion of soil base status, and remobilisation of accumulated sulphur and nitrogen, continue to constrain this recovery in peatland and slow-weathering upland areas of northern England, Wales, southern Scotland and south-east Northern Ireland.

The impacts of atmospheric nitrogen pollution on upland water quality (acidification and eutrophication) is strongly mitigated by plant and microbial uptake and long-term
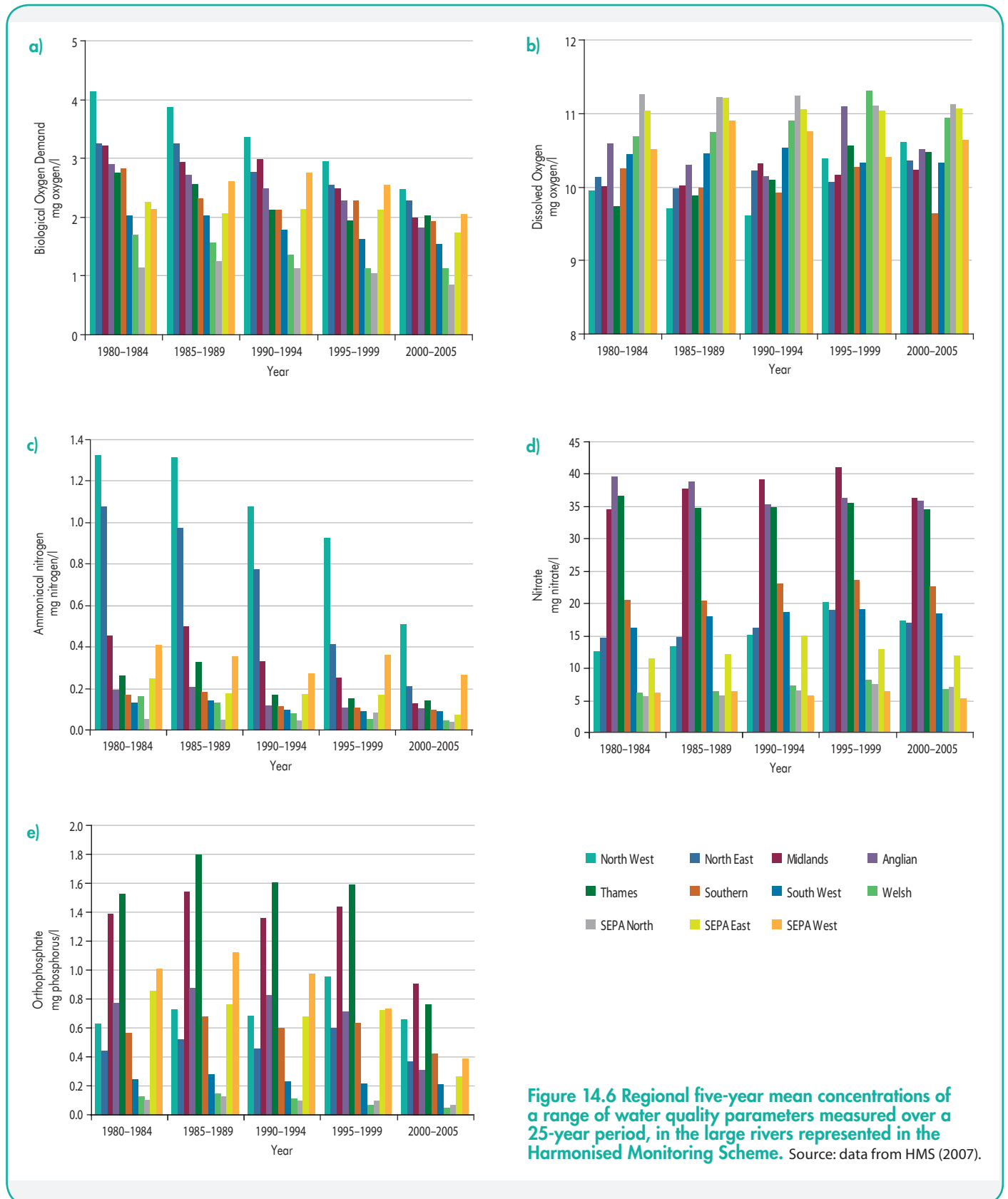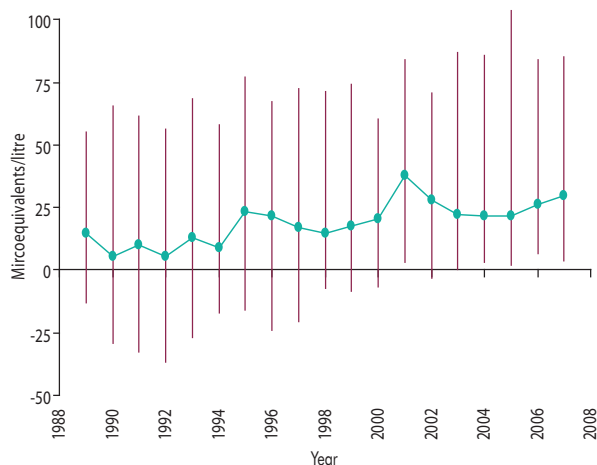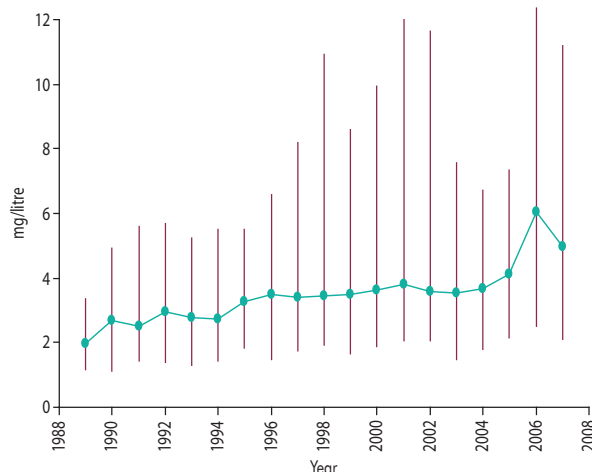

Figure 14.6 Regional five-year mean concentrations of a range of water quality parameters measured over a 25-year period, in the large rivers represented in the Harmonised Monitoring Scheme. Source: data from HMS (2007).

a) Acid Neutralising Capacity

b) Dissolved Organic Carbon

Figure 14.7 Median and 10th–90th percentile range of annual mean concentrations of a) Acid Neutralising Capacity, and b) Dissolved Organic Carbon in the 22 upland lakes and streams of the UK Acid Waters Monitoring Network. Source: Evans *et al.* (2005b); www.ukawmn.ucl.ac.uk.

soil accumulation. With nitrogen emissions expected to remain high, the sustainability of this ecosystem service is uncertain, as semi-natural ecosystems become more nitrogen-enriched (RoTAP 2011). Widespread post-war upland conifer afforestation has had detrimental impacts on water quality, amplifying the effects of acid deposition, and increasing nitrate and sediment losses (Reynolds 2004).

Dissolved organic carbon increases are believed to be linked to recovery from acidification, via increases in SOM solubility (Monteith *et al.* 2007), although climatic factors may also have contributed. Moorland burning and peatland drainage appear to be locally significant (Yallop *et al.* 2010). With atmospheric deposition stabilising at lower levels, climate and land management may be the main determinants of future DOC change. Particulate organic carbon levels are strongly affected by land management; peat erosion in the South Pennines is believed to have been triggered by a combination of historic air pollution, overgrazing and wildfire.

14.9.3 Consequences of Change

Water quality regulates many ecosystem services, and degraded water quality can lead to a breakdown in their function. Eutrophication causes excessive accumulation in the biomass of aquatic plants or algae, resulting in increased treatment costs for downstream users, loss of amenity value due to toxic algal blooms, and the physical blocking of waterways. Under certain conditions, breakdown of accumulated organic matter can lead to deoxygenation of the water, which kills associated fish. Organic pollutants have the same impact, but incidences of this are becoming less frequent. High levels of FIOs lead to increased health risks associated with bathing waters and shellfish beds, which must ultimately be closed as a result. Rising water temperatures may affect ecosystem structure, resulting in the loss of cold water species while favouring the increased success of potentially invasive species.

Over recent decades, acidification led to widespread ecological damage, including biodiversity and fisheries loss

resulting in negative consequences for recreational and provisioning services. This damage is now being reversed in many areas. Rising DOC levels are affecting drinking water provision, adding to treatment costs, and having potential health risks due to formation of halogenated organic compounds during treatment (Singer 1999). Other consequences include the transport of contaminants from soils in soluble organic complexes, and reduced light penetration into the water column, reducing the depth of macrophyte growth but protecting aquatic invertebrates from ultra-violet radiation. Increased particulate export from eroding peatlands also contributes to heavy metal mobilisation (Rothwell *et al.* 2005), and to sediment accumulation and decreased water storage capacity in reservoirs (Yeloff *et al.* 2005).

14.9.4 Options for Sustainable Management

Recent research has indicated that agriculture may be responsible for about 70% of nitrates and 40% of phosphates in UK waters, for 75% of the sediment load to at-risk rivers, and for 36% of bathing beaches and 75% of shellfish waters failing water quality targets (Defra 2009h). The potential for enhanced management of water quality through better land management is thus high, with freshwater-oriented agri-environment schemes, such as the Catchment Sensitive Farming Initiative, providing an implementation mechanism. Specific measures to enhance ecosystem mitigation of diffuse pollutants include: riparian protection and reduced stocking densities; restrictions on timing and loading of fertiliser and manure applications (e.g. within Nitrate Vulnerable Zones); buffer strips to reduce runoff of nutrients and sediment; and the planting of shelter belts to increase water infiltration rates (Kay *et al.* 2007, 2008a, 2010). Constructed wetlands, for example on farms, have potential to purify or de-toxify point source as well as diffuse inputs. Within the drainage network system itself, management options include: riparian wetland restoration (including saltmarshes in the intertidal zone); restoring

channelised rivers to more natural courses in order to increase water residence time and in-channel retention; floodplain reconnection to increase overbank retention; and riparian shading to reduce rates of phytoplankton growth (Withers & Jarvie 2008).

Opportunities also exist to enhance ecosystem regulation of upland water quality. Transition from clear-fell to continuous-cover forestry, riparian protection and improved harvesting practice can all reduce sediment losses, nitrate peaks and other detrimental water quality changes downstream of plantations (Reynolds 2004). In peatlands, reduced stocking levels, limited burning and, in some areas, active restoration can minimise suspended sediment losses. The re-establishment of peat-forming vegetation in degraded areas can also enhance sulphur and nitrogen retention (Bonn et al. 2010). The blocking of peat drains and reduced burning may also reduce DOC losses, although significant uncertainty remains in this area (Armstrong et al. 2010).

The relationship between river flows and water quality will be of critical importance over the next 50 to 100 years due to a combination of climate change predictions of longer low-flow periods punctuated by intensive storm events, projected population growth, increasing water consumption per head, and increased demands for food and bioenergy production. These changes could put unprecedented pressures on the capacity of UK river systems to regulate water quality via dilution, particularly in the south and east. Urban drainage systems are also vulnerable to extreme rainfalls, such as organic pollutant discharges from combined sewage overflows (Kay et al. 2008b; Stapleton et al. 2008). Management of rural and urban land to maximise water infiltration rates will have benefits both in terms of flood mitigation and for maintaining low flows, in order to reduce pollutant pressures on aquatic ecosystems and water supplies.

14.9.5 Knowledge Gaps

Most UK water quality monitoring takes place in large rivers, downstream of human habitations. This may be appropriate for identifying gross diffuse and point source pollution from agricultural and urban areas, but provides limited information on sensitive environments such as upland streams and oligotrophic lakes. The HMS network largely comprises tidal limit sites, providing large-scale integration and flux estimates. Monitoring at this scale, however, is less useful for process understanding, especially relating to linkages between terrestrial and freshwater ecosystems. A recent study showed that 90% of the total river length in Sweden lay within (sub)catchments of <15 km^2 (the minimum threshold for monitoring) and concluded that a major component of the aquatic ecosystem was being overlooked (Bishop et al. 2008). In the UK, monitoring of upland headwater catchments is limited to the UK Acid Waters Monitoring Network, Environmental Change Network, and a number of individual catchment studies. Small lowland catchments are also poorly represented in national monitoring schemes, as are relatively unaffected standing waters.

Routine monitoring under-represents water quality extremes associated with high-flow events. These are responsible for most sediment, phosphorus and microbial pollutant transport into rivers, lakes and estuaries, as well as acid episodes in headwater streams. As such, they have high ecological impacts. The development and deployment of a coordinated network of continuous water quality sensors would dramatically improve understanding of these short-duration processes, which are likely to become more frequent and/or severe under a changing climate. Future changes in

Table 14.21 Interactions between regulating services, other ecosystem services, habitat types and cross-cutting issues. The shade of colour indicates the level of interaction, from dark indicating strong interaction, mid-tone = moderate interaction, to light = limited interaction; blank indicates the interaction is not of particular relevance.

	Climate regulation	Hazard regulation	Disease and pest regulation	Pollination	Noise regulation	Soil quality	Air quality	Water quality
UK NEA Broad Habitat types								
Mountains, Moorlands and Heaths								
Semi-natural Grasslands								
Enclosed Farmlands								
Woodlands								
Freshwaters—openwaters, wetlands and floodplains								
Urban								
Marine and Coastal Margins								
Ecosystem services								
Cultural services								
Supporting services								
Provisioning services								
Cross cutting issue								
Biodiversity								

water temperature are also likely to impact significantly on water quality, but current data are patchy and largely limited to spot-sampling. More formal monitoring may be required to manage extreme events, ideally linked to continuous water quality sampling.

Scientific uncertainties persist in relation to the capacity of terrestrial ecosystems to buffer water quality, including microbial and nutrient retention in wetlands and riparian strips, retention of atmospheric nitrogen by semi-natural ecosystems, and controls on DOC leaching. Understanding of the complex mechanisms by which climate change will impact on water quality in rivers and lakes is incomplete.

Emerging water quality issues include the discharge from the human population of organic and other micro-contaminants, such as synthetic nanoparticles (Ju-Nam & Lead 2008), from sources including pharmaceuticals, and personal care and cleaning products. While not necessarily toxic, these may be ecologically disruptive. For example, up to a third of wild fish living in proximity to sewage effluents have been shown to have endocrine disruption effects (Jobling *et al.* 2006) associated primarily with the contraceptive pill, ethinyloestradiol. Nanosilver, which is used in suntan lotions and clothing, has antimicrobial properties, and so, is treated as a pesticide in the United States (Bardsley *et al.* 2009). The ecological impacts of micro-contaminants are highly variable, and in many cases uncertain, but exposure of British surface waters to such chemicals is probably higher than in most other European nations (Williams *et al.* 2009). At present there is no system in place to identify, monitor or detect the effects on wildlife of organic micro-contaminants, and there is little information available on how these pollutants may be cycled within, or removed from, aquatic ecosystems.

14.10 Trade-offs and Synergies

A number of regulating services either act as final ecosystem services or contribute significantly to final ecosystem services, whilst others are primary or intermediate ecosystem services. There are strong interactions between different regulating services (e.g. water, soil and air quality), and between the regulating services and other ecosystem services. Whilst some regulating services have greater relevance to some habitats types more than others, collectively, regulating services underpin many of the goods and services provided by all habitats. The interactions between the different regulating services and other ecosystem services and habitat types are given in **Table 14.21**.

For water, air and soil quality, human well-being and numerous cultural services are also supported through high quality environments including physical health, mental health, ecological knowledge, cultural heritage and mediated natures, and aesthetic and inspirational.

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Appendix 14.1 Approach Used to Assign Certainty Terms to Chapter Key Findings

This chapter began with a set of Key Findings. Adopting the approach and terminology used by the Intergovernmental Panel on Climate Change (IPCC) and the Millennium Assessment (MA), these Key Findings also include an indication of the level of scientific certainty. The 'uncertainty approach' of the UK NEA consists of a set of qualitative uncertainty terms derived from a 4-box model and complemented, where possible, with a likelihood scale (see below). Estimates of certainty are derived from the collective judgement of authors, observational evidence, modelling results and/or theory examined for this assessment.

Throughout the Key Findings presented at the start of this chapter, superscript numbers and letters indicate the estimated level of certainty for a particular key finding:

1. *Well established:* high agreement based on significant evidence
2. *Established but incomplete evidence:* high agreement based on limited evidence
3. *Competing explanations:* low agreement, albeit with significant evidence
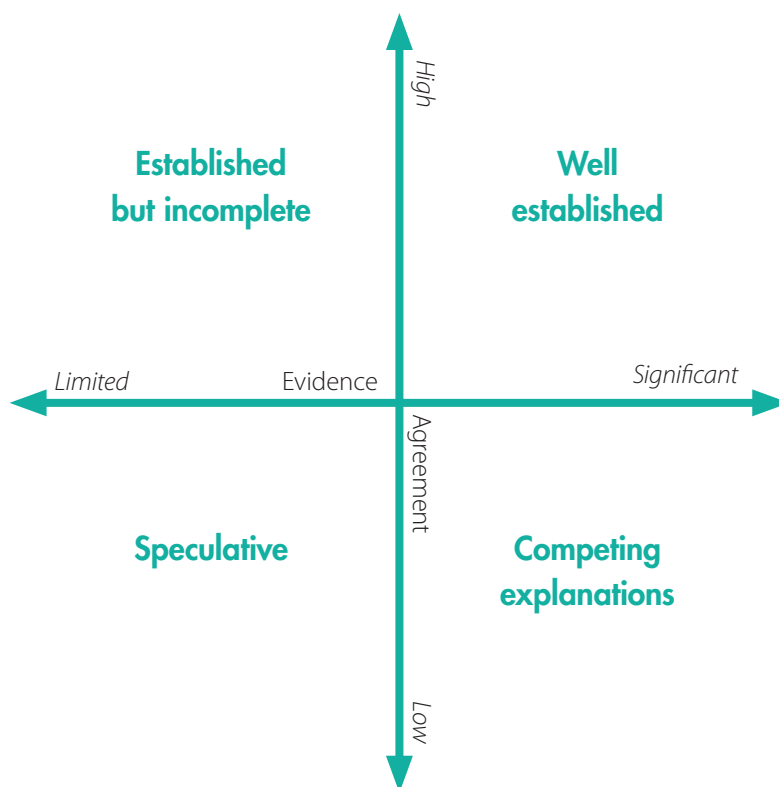
4. *Speculative:* low agreement based on limited evidence
a. *Virtually certain:* >99% probability of occurrence
b. *Very likely:* >90% probability
c. *Likely:* >66% probability
d. *About as likely as not:* >33–66% probability
e. *Unlikely:* <33% probability
f. *Very unlikely:* <10% probability
g. *Exceptionally unlikely:* <1% probability

Certainty terms 1 to 4 constitute the 4-box model, while a to g constitute the likelihood scale.