Chapter 4:
Biodiversity in the Context of Ecosystem Services

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Key Findings*

The term 'biodiversity' describes the diversity of life on Earth. Diversity can occur at a number of levels of biological organisation, from genes, through to individuals, populations, species, communities and entire ecosystems[1].

[1] *well established*

Biodiversity underpins all ecosystem services. Biodiversity plays a wide range of functional roles in ecosystems and, therefore, in the processes that underpin ecosystem services[1]. Examples range from the roles bacteria and fungi play in nutrient cycles which are fundamental processes in all ecosystems, to particular animal groups, such as birds and mammals, which are culturally important to many people. Ecosystem functions are more stable through time in experimental ecosystems with relatively high levels of biodiversity[2]; and there are comparable effects in natural ecosystems[c]. Taken together, this evidence shows that, in general terms, the level and stability of ecosystem services tend to improve with increasing biodiversity.

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

Biodiversity plays a wide range of roles in UK ecosystem services. All twelve of the ecosystem services that are important in a UK context are underpinned by a range of biodiversity groups. The number of biodiversity groups playing an important role varies between ecosystem services: water quantity (3/17 of biodiversity groups); socially valued landscapes and waterscapes (6/17 groups); crops, plants, livestock and fish (11/17 groups); and wild species diversity (all 17 groups). The role of different biodiversity groups varies between ecosystem services. Microorganisms, fungi and plants play a role in underpinning all provisioning and regulating services; vertebrate groups contribute to all cultural services, but they only play an important role in 30% (3/10) of the provisioning and regulating services.

Biodiversity is a key component of multifunctional ecosystems. The importance of managing ecosystems to provide multiple services and associated values (so-called 'multifunctional ecosystems') is becoming increasingly recognised both globally and in the UK. The sensitivity of UK ecosystem services to changes in a range of biodiversity groups implies that achieving this multifunctionality will require management measures to support a wide range of biodiversity groups.

Significant biodiversity loss has been documented in the UK over the last 50 years, but monitoring data for a number of biodiversity groups is poor, precluding an assessment of status and trends. The quality of monitoring data in the UK varies between biodiversity groups. For some biodiversity groups, such as marine plankton, land plants, some invertebrate groups, fish, birds and mammals, national-scale data on abundance and range exist for a time-series of 10–20 years. These datasets show clear patterns of biodiversity change. The quality of monitoring data across UK biodiversity groups increases in relation to their cultural importance. As a result, there are only limited data available on several biodiversity groups, such as microorganisms and fungi, which underpin provisioning and regulating services, precluding an assessment of their status and trends.

* 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 indicate the uncertainty term assigned to each finding. Full details of each term and how they were assigned are presented in Appendix 4.2.

Relating changes in UK biodiversity to changes in ecosystem services can be problematic due to a lack of data on associated values and benefits. Interpreting the impact of even well-established trends in UK biodiversity on associated ecosystem services can be problematic where data on values and benefits are lacking. For example, we lack quantitative data on cultural services, so we are currently unable to assess the magnitude of changes in cultural services associated with well-established changes in bird populations. In contrast, specific, well-established biodiversity trends linked to provisioning and regulating services can have clear implications for service provision. For example, declines in the abundance of commercially important marine and freshwater fish species lead directly to a reduction in the output of provisioning services.

Land use change and pollution have been the major drivers of change across biodiversity groups in the UK. Land use change is considered a significant driver of change across all UK biodiversity groups associated with terrestrial and freshwater ecosystems, and for marine groups affected by activities on land. For example, recent evidence suggests that about 67% of 333 farmland species (broadleaved plants, butterflies, bumblebees, birds and mammals) were threatened by agricultural intensification in the year 2000. Pollution impacts reflect a range of human activities including diffuse pollution from agriculture, point source pollution from urban ecosystems, and air pollution (e.g. acid rain).

There is a cultural divide among biodiversity groups and associated ecosystem services in the UK. On one side of this divide are culturally important biodiversity groups; on the other side are biodiversity groups that underpin provisioning and regulating services. For several culturally important biodiversity groups, status and trends are well-established, but data on associated cultural services are frequently lacking. This makes it difficult to quantify the impact of biodiversity change on cultural services. For provisioning and regulating services, quantitative data on changes in the services themselves are often available, but status and trend information for associated biodiversity groups is considered poor. This makes it difficult to understand the role biodiversity plays in changes in associated provisioning and regulating services. Bridging this cultural divide represents a major research and policy challenge.

4.1 Background

Charles Darwin famously described the diversity of life on Earth as "endless forms most beautiful". Over the subsequent 100 years or so, it has become increasingly apparent that human activities have caused, and continue to cause, significant loss of this diversity. This realisation culminated in the Convention on Biological Diversity (CBD) in 1992, which established policies for the conservation of biodiversity, the sustainable use of its components, and the fair and equitable sharing of benefits arising from biodiversity. Subsequently, there has been considerable debate about appropriate indicators that can be used to measure the health of biodiversity (Balmford *et al.* 2003; Green *et al.* 2005) and a number of countries, including the UK, have adopted biodiversity targets and indicators to report on biodiversity status and trends (Gregory *et al.* 2004). These national initiatives complement global targets and indicators, which have recently shown that, despite commitments to halt biodiversity loss by 2010, significant biodiversity loss continues (Butchart *et al.* 2010).

It is widely accepted that biodiversity plays a wide range of key functional roles within terrestrial, freshwater and marine ecosystems (Hooper *et al.* 2005; Raffaelli 2006; Worm *et al.* 2006; Palumbi *et al.* 2009). Nevertheless, it was not until the Millennium Ecosystem Assessment (MA) in 2005 that these functional roles were viewed holistically in the context of ecosystem services and benefits linked to human well-being (MA 2005). The MA recognised the critical roles played by biodiversity in underpinning ecosystem services (**Figure 4.1**). Subsequent work, such as the European Academies Science Advisory Council's (EASAC) report on Ecosystem Services and Biodiversity in Europe and the report, Reviewing the Economics of Biodiversity Loss: Scoping the Science, produced as part of The Economics of Ecosystems and Biodiversity (TEEB) project, has attempted to be more explicit about how biodiversity underpins the delivery of ecosystem services, and considers the potential consequences of biodiversity loss for future service delivery (Balmford *et al.* 2008; TEEB 2008; EASAC 2009; TEEB 2009). While we often have a broad understanding of which biodiversity groups are important in underpinning specific ecosystem services, such assessments are frequently hampered by a critical lack of quantitative data on biodiversity and ecosystem service relationships at the scales (spatial and temporal) typical of real-world ecosystems (Balmford & Bond 2005; Kremen 2005).

Theoretically, there are a number of potential relationships between biodiversity and ecosystem services (**Figure 4.2a**). Describing these patterns is the key to determining the consequences of biodiversity loss for ecosystem services. While there has been considerable research on the relationships between biodiversity and ecosystem function over the last 20 years (Hooper *et al.* 2005; Raffaelli 2006), much of this work has limitations in terms of understanding real-world ecosystems (Srivastava & Vellend 2005). This is because studies have typically been undertaken on small-scales and within highly simplified experimental ecosystems. Studies using 'model' ecosystems are valuable in exploring the functional roles of biodiversity, but how they relate to biodiversity and ecosystem change in the real world is less clear (Kremen 2005). As a result,

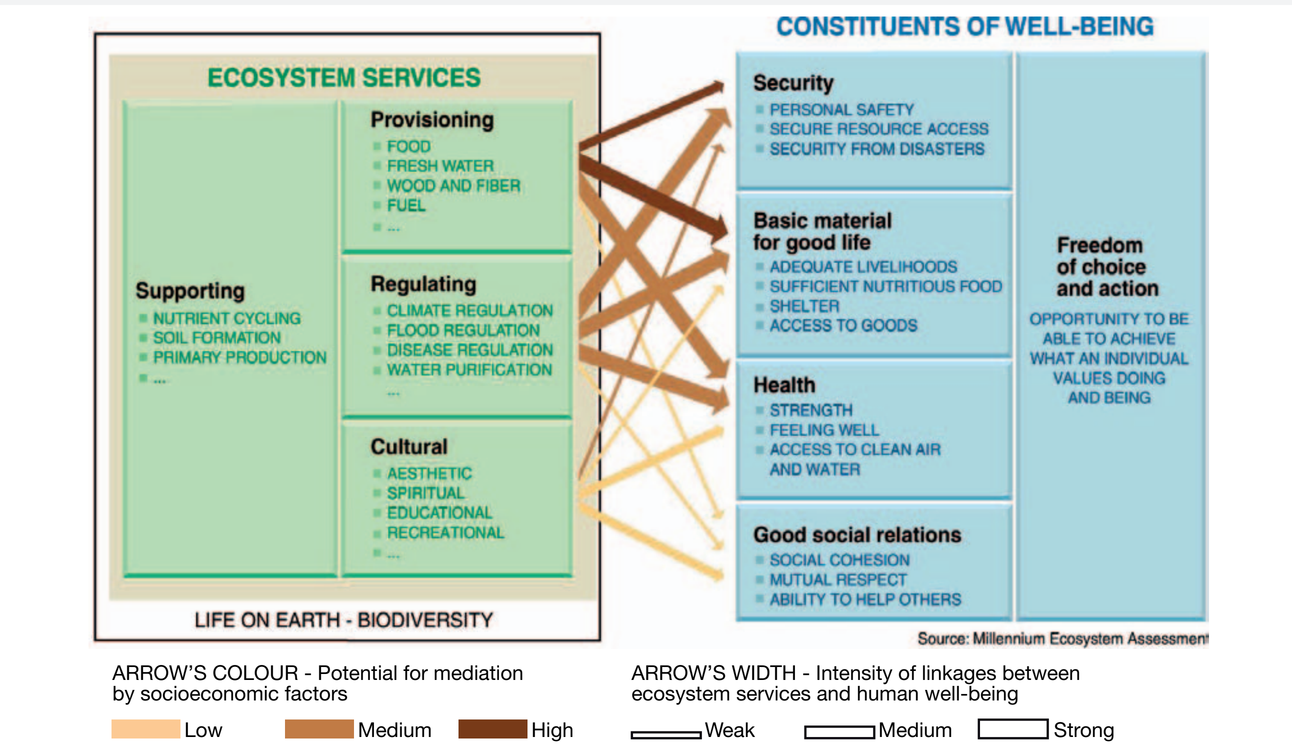


Figure 4.1 The relationships between biodiversity, ecosystem services and human well-being. Source: reproduced from the MA (2005).

there are increasing calls among the scientific community to focus research explicitly on understanding relationships between biodiversity and ecosystem services in the context of real-world ecosystem change (Srivastava & Vellend 2005; Raffaelli 2006); thus insights from natural systems are beginning to accumulate (Benayas *et al.* 2009).

Against this background, it is perhaps not surprising that our understanding of the quantitative links between biodiversity and ecosystem services is, at present, generally rather poor (Kremen 2005), and is limited to a few well-understood case studies such as crop pollination services (Kremen *et al.* 2002) and disease regulation (Keesing *et al.* 2006). There are two important issues relating to this understanding. Firstly, the functional component of biodiversity needs to be identified. It is possible that a service is related to an aspect of diversity *per se* (e.g. species diversity); alternatively, the service may depend on a specific functional group or even an individual species that plays a specific functional role. The functional components of biodiversity may also vary between types of ecosystem service (Diaz *et al.* 2007). Secondly, the data available relating biodiversity to a particular ecosystem service are often relatively sparse (**Figure 4.2a**). Therefore, it is possible to show that a specific ecosystem service is sensitive to changes in a particular biodiversity group, but it is often the case that there is just not enough information available to describe the form of the relationship. As a result, the available evidence is good enough to be able to demonstrate that biodiversity matters to the provision of ecosystem services, but it is often not good enough to allow us to distinguish services that are sensitive to even small levels of biodiversity loss from those that are more resilient to biodiversity loss (**Figure 4.2a**).

There is a general consensus in the literature that biodiversity enhances the stability of ecosystems (Hooper *et al.* 2005). This is believed to occur because increasing biodiversity also increases functional diversity, thereby buffering ecosystem processes against temporal (**Figure 4.2b**) or spatial perturbation (Loreau *et al.* 2003). These concepts are important in the context of ecosystem services because they imply that, as biodiversity is lost from an ecosystem, service provision is not only likely to decrease to some extent (**Figure 4.2a**), but may also get more variable in space or time (**Figure 4.2b**). As a result, biodiversity has a potentially important 'insurance' role to play in maintaining service provision in the face of environmental change.

4.2 Biodiversity in the Context of the UK National Ecosystem Assessment

The general issues discussed in the previous section have important implications for how we consider biodiversity within the UK NEA. The UK has, perhaps, the most comprehensive data on biodiversity status and trends of any country in the world, but these data are not routinely linked

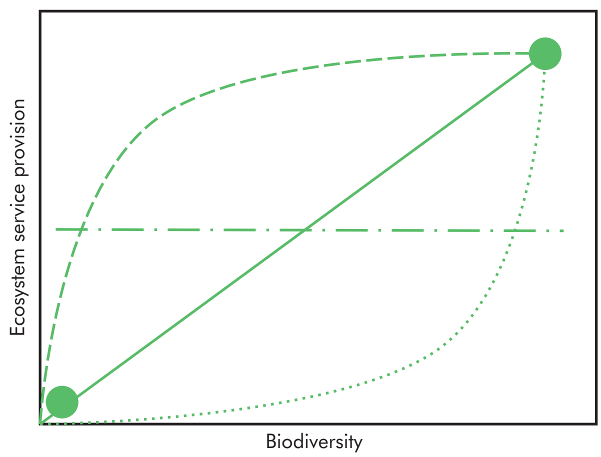


Figure 4.2a Theoretical relationships between biodiversity and an ecosystem service. The dashed line shows that the ecosystem service is resilient to moderate levels of biodiversity loss; whereas the dotted line shows that the service is very sensitive to even small levels of biodiversity loss. The solid line is intermediate between these two. The dashed and dotted line illustrates the case in which an ecosystem service is insensitive to biodiversity change. The green dots illustrate the type of data that are typically available. These data show that biodiversity loss reduces the provision of the ecosystem service, but are too sparse to describe the relationship in any detail.

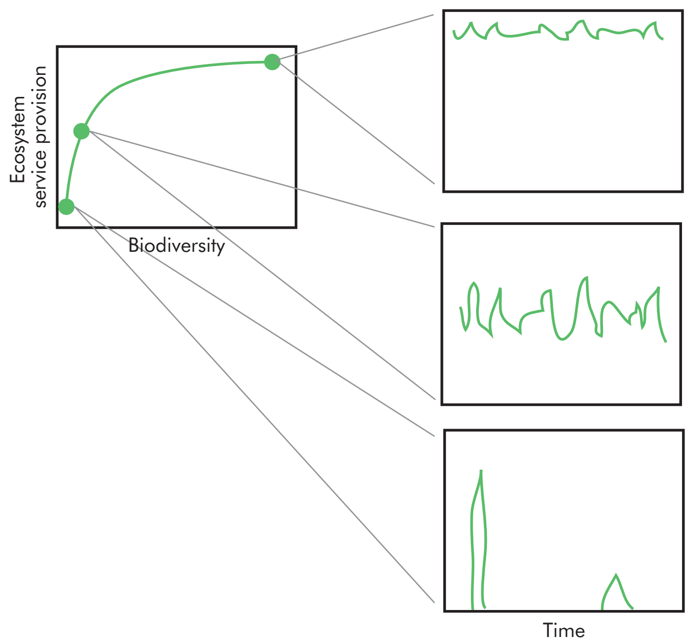


Figure 4.2b The stability of ecosystem services through time in relation to biodiversity. The left hand panel illustrates the theoretical relationship between biodiversity and the provision of an ecosystem service. The right hand panels illustrate how the provision of the service may become more variable through time as biodiversity decreases.

to ecosystem services. For example, it is well known that pollinating insects play a crucial role in providing pollination services to agricultural crops (Klein *et al.* 2007; Zhang *et al.* 2007). While we have evidence of pollinator losses in the UK (Biesmeijer *et al.* 2006), we have a very limited understanding of the consequences of these losses for pollination services, or of how environmental change is likely to impact on pollination systems. Consequently, a £10 million research programme is currently underway to address these knowledge gaps (see www.lwec.org.uk/activities/insect-pollinators-initiative). This is relevant in the context of the UK NEA's consideration of biodiversity because pollination is one of the best understood biodiversity-ecosystem function-ecosystem service relationships. Taken together, this significant lack of evidence means we are currently unable to comprehensively quantify the relationships between UK biodiversity and the ecosystem services it supports (**Figure 4.2a, b**).

For these reasons, we have qualitatively assessed (low, medium or high) the importance of a range of biodiversity groups in underpinning the final ecosystem services being covered by the UK NEA, with the aim of identifying key biodiversity groups associated with each final ecosystem service. This assessment is based on the premise that, while we often understand that a particular ecosystem service (e.g. pollination) is likely to be sensitive to changes in specific biodiversity groups (e.g. pollinating insects), we are not able to quantify this sensitivity (Section 4.1). We have also reviewed and synthesised the available status and trend information for each of these biodiversity groups, and discussed the linkages between the status and trend data and ecosystem services. Lastly, we synthesised the available information on drivers of biodiversity change to identify important factors that may modify biodiversity in the UK and the ecosystem services it supports.

To undertake this assessment it was necessary to define 'biodiversity'. The CBD defines biodiversity as "the variability among living organisms from all sources, including, 'inter alia', terrestrial, marine, and other aquatic ecosystems, and the ecological complexes of which they are part: this includes diversity within species, between species and of ecosystems". This definition is less than ideal from the perspective of ecosystem services because, as noted in the previous section, diversity *per se* may have only a limited effect on specific ecosystem services. The scientific debate about how best to define biodiversity must also recognise practical constraints imposed by available biodiversity data. In the UK, biodiversity data tends to relate to taxonomic groups distinguished by specific monitoring programmes (e.g. www.nbn.org.uk), which, in turn, provide the data that are used to report on status and trends. While it would be possible, at least in principle, to redefine the biodiversity groups recognised in the UK in terms of their functional roles in ecosystem services, such a task would inevitably be problematic due to the need to combine datasets from a range of monitoring programmes that employ different census methods. This would be a significant undertaking and beyond the scope of this chapter. As a result, this chapter has taken a pragmatic approach by adopting the taxonomic groups recognised by UK monitoring programmes to define UK biodiversity (**Table 4.1**).

These biodiversity groups form the basis of the assessment reported in this chapter. The assessment itself was conducted by a team of 35 scientists with specific expertise in the range of biodiversity groups involved (**Appendix 4.1**). Before reporting the assessment, we briefly outline in the following section how biodiversity fits into the conceptual framework being used by the UK NEA.

Table 4.1 Biodiversity groups distinguished in the UK NEA.

Biodiversity group	Definition
Microorganisms	Bacteria and Archaea, formerly grouped as the prokaryotes, and the single-celled Eukaryotes.
Fungi and Lichens	Mycetozoan (e.g. Myxomycota) and Heterokontophytan (e.g. Oomycota) species; lichenised-fungi include Ascomycetes ('cup'-fungi) and Basidiomycetes.
Phytoplankton	Photoautotrophic microorganisms found in aquatic ecosystems, e.g. diatoms, cyanobacteria, dinoflagellates and coccolithophores.
Macroalgae	Multicellular eukaryotic algae belonging to one of three main groups; red algae (Rhodophyta), green algae (Chlorophyta) and brown algae (Phaeophyceae).
Bryophytes	Liverworts (Marchantiophyta), mosses (Bryophyta) and hornworts (Anthocerotophyta).
Seagrasses	Two species of seagrass; the primarily subtidal *Zostera marina* (eelgrass) and the intertidal *Zostera noltii* (dwarf eelgrass).
Land plants	All vascular plants: Lycopods, Isoetes and Selaginella, ferns and horsetails, conifers (Gymnosperms), and all flowering plants (Angiosperms)—trees, shrubs, herbaceous plants and grasses. The majority are land plants, but some occur in freshwater, brackish or marine habitats.
Invertebrates	All marine, freshwater and terrestrial invertebrates (e.g. annelids, crustaceans, molluscs, arthropods, echinoderms).
Fish	All marine and freshwater fish.
Amphibians	Frogs, toads and newts.
Reptiles	Snakes, lizards and marine turtles.
Birds	Land and seabirds.
Mammals	Land mammals, cetaceans and pinnipeds.

4.3 Biodiversity and the Conceptual Framework

We illustrate how biodiversity fits into the conceptual framework being used by the UK NEA in **Figure 4.3**. Biodiversity can potentially play a role in the primary and intermediate processes that underpin final ecosystem services, and it can also play a role in the final ecosystem services themselves. Our example considers a fruit crop pollinated by insects. Biodiversity is part of the ecosystem processes that provide pollination services to fruit crops, and it is also part of the fruit crop itself (the final ecosystem service) because wild and domesticated plants provide the raw material from which crop varieties are derived. Clearly,

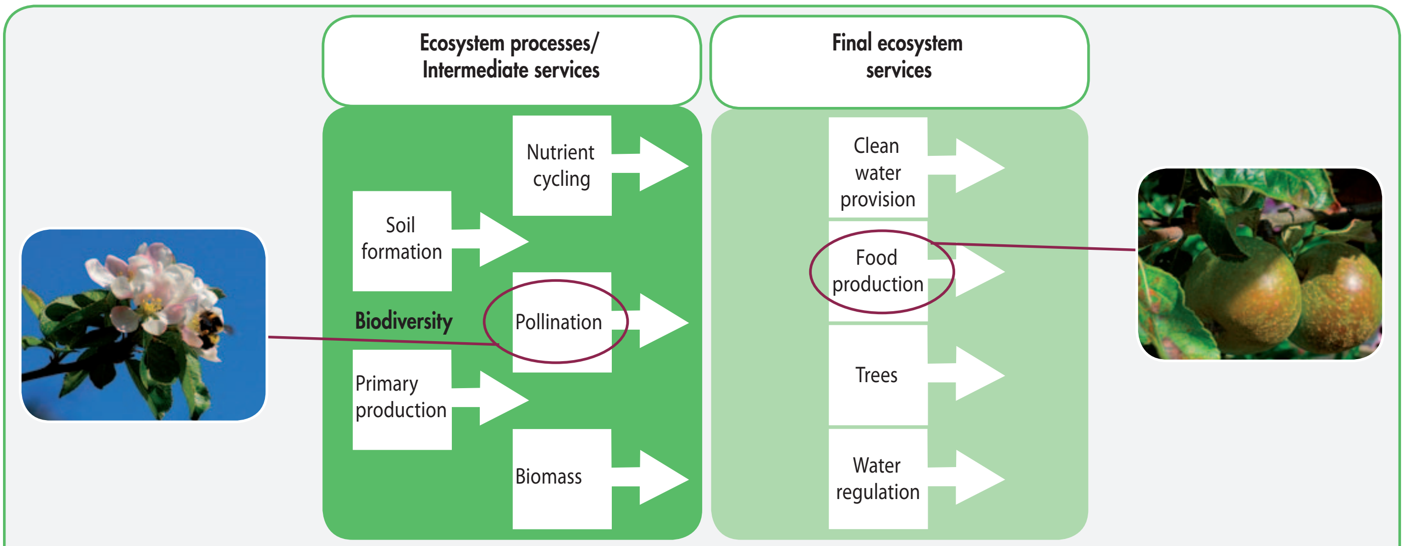


Figure 4.3 The roles of biodiversity within the conceptual framework being used by the UK NEA. The figure illustrates that biodiversity can have a role in the processes underpinning the final services (in this case pollination) or be part of the final service itself (in this case a fruit crop). *Photo sources: Bumblebee, Bombus hortorum, pollinating apples by John Fergusson; Braeburn apples by John Thurm, available under a Creative Commons Attribution-NonCommercial-ShareAlike license.*

the networks of ecological interactions that underpin a final ecosystem service, such as a fruit crop, are more complex than implied by our example. Arguably, all of the primary and intermediate processes play some role (**Figure 4.3**), and biodiversity is likely to play a key role in a number of these. Nevertheless, the important point is that when we talk about biodiversity 'underpinning' the delivery of ecosystem services it is either in the context of biodiversity being part of primary or intermediate processes, or part of the final service itself; often it will be both.

4.4 The Role of Biodiversity in UK Ecosystem Services

In broad terms, we know which biodiversity groups play potentially important roles in UK ecosystem services, but we lack quantitative data that would allow us to link current biodiversity status and trend data with the delivery of ecosystem services. For this reason, we have qualitatively assessed the importance of different biodiversity groups using expert opinion and by adopting a similar approach to that used by the EASAC study (EASAC 2009). The EASAC study assessed the importance of biodiversity using a simple scale of low, medium and high, which we have also adopted. While our approach is similar to EASAC's, it has been specifically tailored to the UK context in terms of the biodiversity groups (**Table 4.1**) and ecosystem services (**Table 4.2**) being assessed. Experts for each biodiversity group were asked to assess the importance of their biodiversity group in underpinning each final ecosystem service being considered in the UK NEA using a simple scale of low, medium or high. This assessment did not consider the precise role played by biodiversity, but simply whether a particular group was considered important irrespective of the details of its role. In this way, we aimed to identify key biodiversity groups associated with each final ecosystem service. Experts were also asked to identify the level of uncertainty in the available evidence.

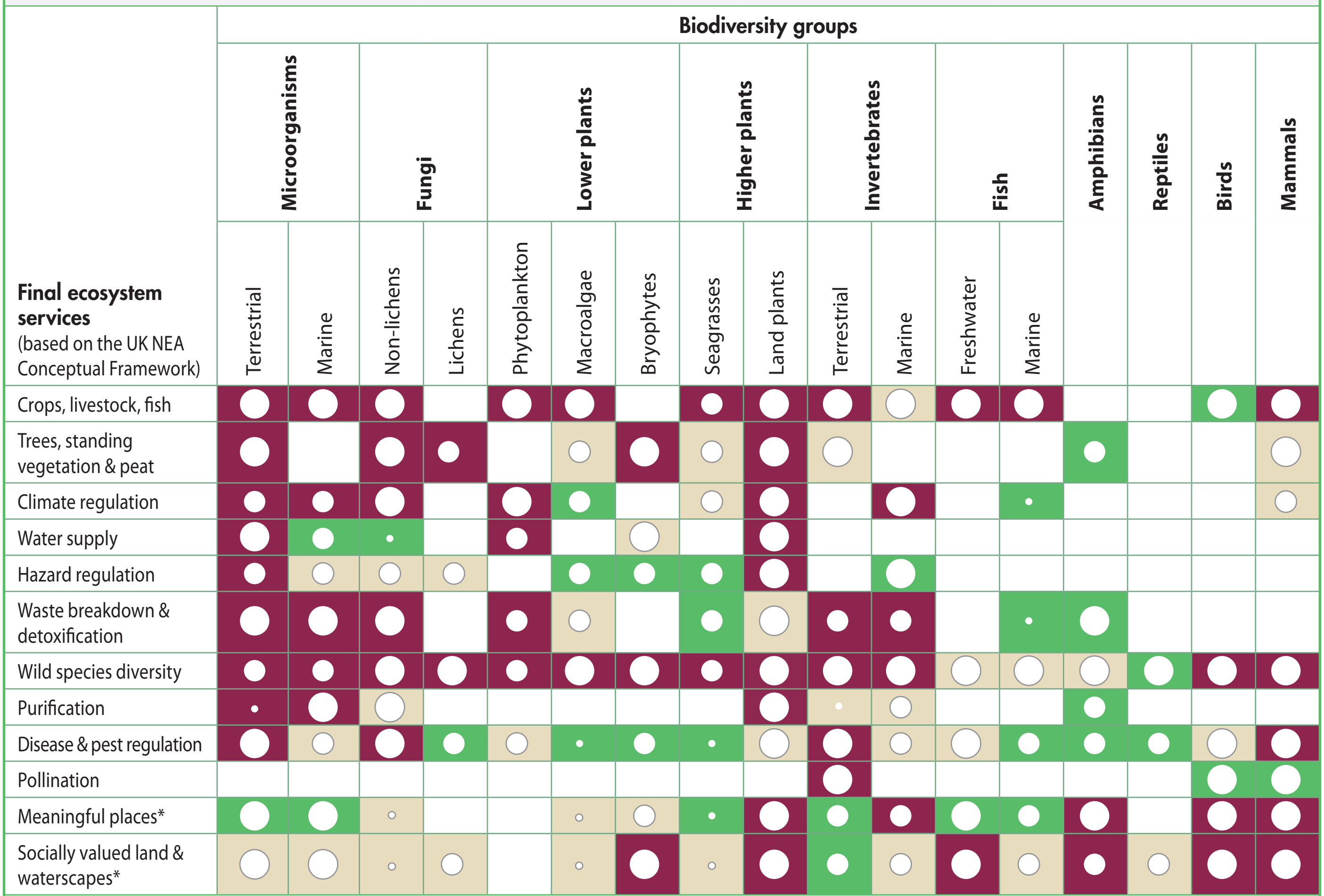
This general concept of importance is being used to qualitatively assess the 'sensitivity' of each ecosystem service to changes in each biodiversity group. Where importance is considered 'high', this should be taken to mean that the particular ecosystem service is relatively sensitive to changes in the specific biodiversity group being assessed; where importance is considered 'low', the particular service is relatively insensitive to changes in the specific biodiversity group being assessed. The concept of importance does not reflect the functional mechanism linking the biodiversity group with a specific ecosystem service. As a result, 'high' importance might reflect sensitivity of a specific ecosystem service to levels of diversity present within a particular biodiversity group; but it might also reflect sensitivity to the presence or abundance of specific functional groups, species or genotypes within a particular biodiversity group. In addition, the concept of importance does not explicitly consider the issue of irreplaceability: the idea that the functional role performed by biodiversity cannot be substituted by an artificial process. It simply provides a basis for comparison across a range of biodiversity groups and ecosystem services irrespective of the functional mechanisms involved.

The results of this assessment are summarised in **Table 4.2**. The rows of the table list the final ecosystem services being covered within the UK NEA; the columns identify the different biodiversity groups. The cells in the table are colour-coded to reflect the degree of importance assigned to each service-biodiversity group combination, ranging from high (maroon) to low (green) importance. The size of the circle in each cell is used to illustrate the level of uncertainty in the available evidence.

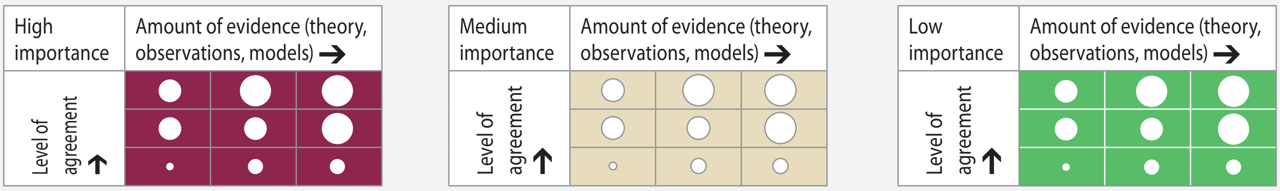
A number of specific points emerge:

- All UK ecosystem services are dependent on biodiversity to some extent.

Table 4.2 The importance of different biodiversity groups in underpinning the final ecosystem services based on expert opinion. Importance is colour-coded: high (maroon), medium (beige), low (green), unimportant on the basis of available evidence (blank). The size of the circle in each cell is used to illustrate the level of uncertainty in the available evidence. Further details are given in Appendix 4.1.



* Note: For the purposes of the Cultural Services chapter (Chapter 16), Cultural services have been combined into 'environmental settings'.



- Over 60% (11/17) of the biodiversity groups assessed play an important role in underpinning the crops, plants, livestock and fish upon which we depend for food.
- Microorganisms, fungi and plants play key roles in provisioning and regulating services.
- Higher plants and animals play key roles in cultural services.

The finding that all UK ecosystem services are sensitive to changes in more than one biodiversity group has important implications for the concept of multifunctional ecosystems and the implementation of an 'ecosystems approach' in the UK. The importance of managing ecosystems to provide multiple services and associated values is becoming increasingly recognised both at an international level (Chan *et al.* 2006; Kareiva *et al.* 2007; Naidoo *et al.* 2008; Norris 2008; Bennett *et al.* 2009; Nelson *et al.* 2009) and in a UK context (Anderson *et al.* 2009). In turn, this recognition is stimulating policy responses to explore how a multifunctional ecosystems approach might work in practice (for example, Natural England's ecosystem pilot projects). The evidence summarised in **Table 4.2** suggests that an important objective of these developments should be the management of UK ecosystems to support biodiversity across a wide range of groups to ensure the provision of a range of ecosystem services. The scientific challenges involved with developing the necessary evidence base are significant, but research programmes are emerging that aim to better understand the functional links between biodiversity and ecosystems services in the context of UK ecosystems (e.g. www.nerc.ac.uk/research/themes/tap/tap-phase2.asp). It will be important for the emerging science in this area to interface appropriately with policy development.

4.5 Biodiversity Status and Trends

4.5.1 The Quality of Monitoring Data in Relation to Ecosystem Services

The quality of biodiversity monitoring data in the UK varies between biodiversity groups. In broad terms, the status and trends data tends to be of a higher quality for biodiversity groups closely associated with the cultural services being assessed by the UK NEA (**Figure 4.3**). To investigate this relationship, the trend information available for each biodiversity group was classified as 'good' (UK-wide data on distribution, abundance and population trends over a 20-year or more time period), 'moderate' (UK-wide data on distribution, but limited data on abundance and population trends due to spatial or temporal coverage), 'patchy' (only localised data available on distribution or trends) or 'poor' (negligible data available on distribution or trends) (details are summarised in **Table 4.3**). The pattern in **Figure 4.3** partly reflects technical difficulties associated with monitoring specific groups associated with provisioning and regulating services, such as microorganisms, and suggests that many long-term monitoring schemes were initiated, at least in part, for cultural reasons. As a result, biodiversity groups associated with provisioning and regulating services are often poorly monitored, hence we have a limited understanding of their status and trends.

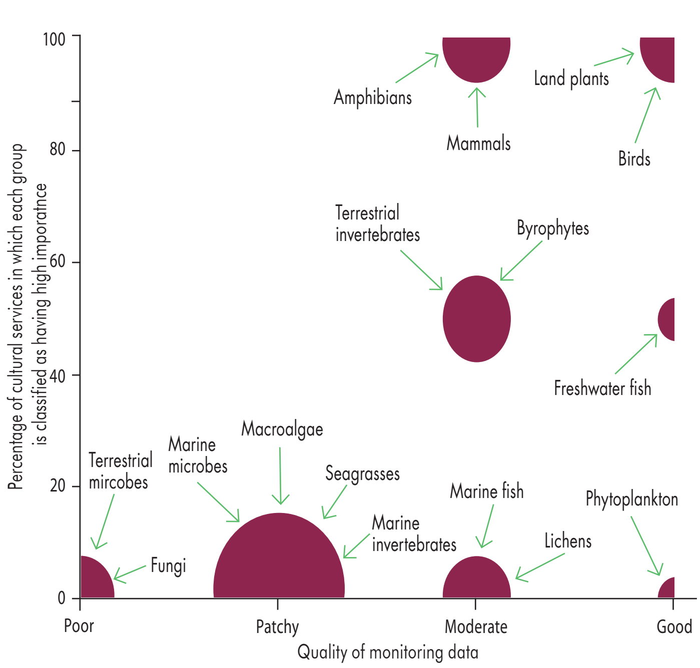


Figure 4.4 The relationship between the quality of monitoring data for each biodiversity group and its role in cultural services. Importance is taken from Table 4.2 in which 'high' importance is coloured maroon. The size of the circle indicates the number of biodiversity groups associated with each point. The relationship can be described by the simple linear model: $y = a+b.x$, in which x takes integer values from 1 (poor) to 4 (good). This model is statistically significant (R^2_{adj} = 0.30, P = 0.016).

4.5.2 Status and Trend Information

The Joint Nature Conservation Committee (JNCC) produces a series of UK Biodiversity Indicators, which includes an assessment of the status and trends of components of biodiversity (JNCC 2010b; **Table 4.4**). In general terms, these indicators show improving or stable trends in species, habitats and protected sites of high conservation priority (indicator groups 3–6) over the last decade, but declining trends among biodiversity groups in the wider environment (indicator groups 1–2). Of the 11 specific indicators in these latter two groups, more than 70% (8/11) have shown declining trends in the recent past.

The JNCC indicators represent only part of the status and trend data available for UK biodiversity. The JNCC's online wildlife statistics database (www.jncc.gov.uk/page-3254) contains more than 7,000 trends from over 4,000 species, while the National Biodiversity Network (NBN) (www.nbn.org.uk/) contains more than 57 million species records. For some biodiversity groups, such as marine plankton, land plants, certain invertebrate groups, fish, birds and mammals, national-scale data on abundance and range exist for a time-series of more than 10–20 years. These datasets show clear patterns of biodiversity change:

- Plankton survey data has documented a northward shift in species diversity in the Atlantic Ocean over the last 20 years (**Figure 4.5a**).
- Atlas data for native land plants show that ranges have, on average, contracted since the 1960s across 1,142 native species.
- Countryside Survey data shows a downward trend in average plant diversity across most habitats between 1978 and 2007 (**Figure 4.5b**), but evidence of increased soil invertebrate abundance in all habitats except arable during the same time period—differences which were largely due to greater mite populations (Emmett *et al.* 2010). However, the Countryside Survey also indicated a small decrease in soil invertebrate biodiversity with the number of broad invertebrate taxa present in samples generally lower in 2007 than in 1998 (Emmett *et al.* 2010).
- Results from the Countryside Survey also illustrate improvements to the diversity of freshwater invertebrates in headwater streams across Great Britain (GB) since 1990; however, in lowland ponds, they may have declined.
- The populations of butterfly species that are specialists of semi-natural habitat have more than halved since 1976 (**Figure 4.5c**).
- Marine fish populations and communities have changed significantly since the 1960s, with exploited populations

Table 4.3 Population trends of wild bird species in different habitats. Source: data from the RSPB, BTO and Defra (2010).

Species group (number of species)	Long-term trend	Short-term trend	
Breeding birds	**1970–2008**	**1998–2008**	**Key drivers**
All species (114)	3%	6%	Multiple and diverse
Seabird species (19)	28%	-5%	Fishery practice and oceanic change
Water and wetland species (26)	1%	9%	Change in agricultural practices
Woodland species (38)	-14%	5%	Change in woodland structure
Farmland species (19)	-47%	-4%	Change in agricultural practices
Urban species (27)	-	11% *	Sympathetic management and food provision
Wintering birds	**1975/1976–2006/2007**	**1996/1976–2006/2007**	
All waterbird species (46)	57%	-6%	Site and species protection and management
Wildfowl species (27)	62%	-9%	
Wader species (15)	44%	-5%	

* English trends 1994–2008.

Table 4.4 Status and trends in components of UK biodiversity. The symbols in the cells of the table indicate the direction of trends: declining (↓), increasing (↑) and stable (=). # denotes data is not available. Note: with the long-term change the baseline year varies between categories, see JNCC (2010a) for details. Source: data extracted from JNCC (2010a).

Status and trends in components of biodiversity		Long-term change	Change since 2000
1a. Population trends of selected species (birds)	Breeding farmland birds	↓	↓
	Breeding woodland birds	↓	↑
	Breeding water and wetland birds	=	=
	Breeding seabirds	↑	↓
	Wintering waterbirds	↑	↓
1b. Population trends of selected species (butterflies)	Semi-natural habitat specialists	↓	=
	Generalist butterflies	=	=
1c. Population trends of selected species (bats)		↓	↑
2. Plant diversity	Arable and horticultural land	↑	↑
	Woodland and grassland	↓	↓
	Boundary habitats	↓	↓
3. UK Priority species		#	↑
4. UK Priority habitats		#	=
5. Genetic diversity	Native sheep breeds	#	=
	Native cattle breeds	#	↑
6. Protected areas	Total extent of protected areas	↑	↑
	Condition of Areas/Sites of Special Scientific Interest	#	↑

declining in abundance and some vulnerable species, such as the common skate (*Raja batis*), disappearing entirely from some areas of their range. Since the early 1990s, there is evidence of population recovery in 10–20% of finfish populations (**Figure 4.5d**).

- Among freshwater fish, there is evidence of significant declines in commercially important species, with the number of young European eels (*Anguilla anguilla*) returning to rivers falling to 1% of historical levels since the 1980s.
- Status and trends among wild bird species varies between habitats. Seabird populations have increased by 28% since 1970, but have decreased (-5%) over the last decade; woodland and farmland populations have both declined (-14% and -47% respectively); whereas urban populations have increased (11%) (**Figure 4.5e**).
- Among 37 UK mammal species, 40% appear to be increasing, 12% declining, and 16% stable, with the remaining 32% being considered data deficient (**Figure 4.5f**).

Across biodiversity groups with adequate data, there is clear evidence of significant biodiversity losses (i.e. range contractions and population declines), together with evidence of population increases in certain species and

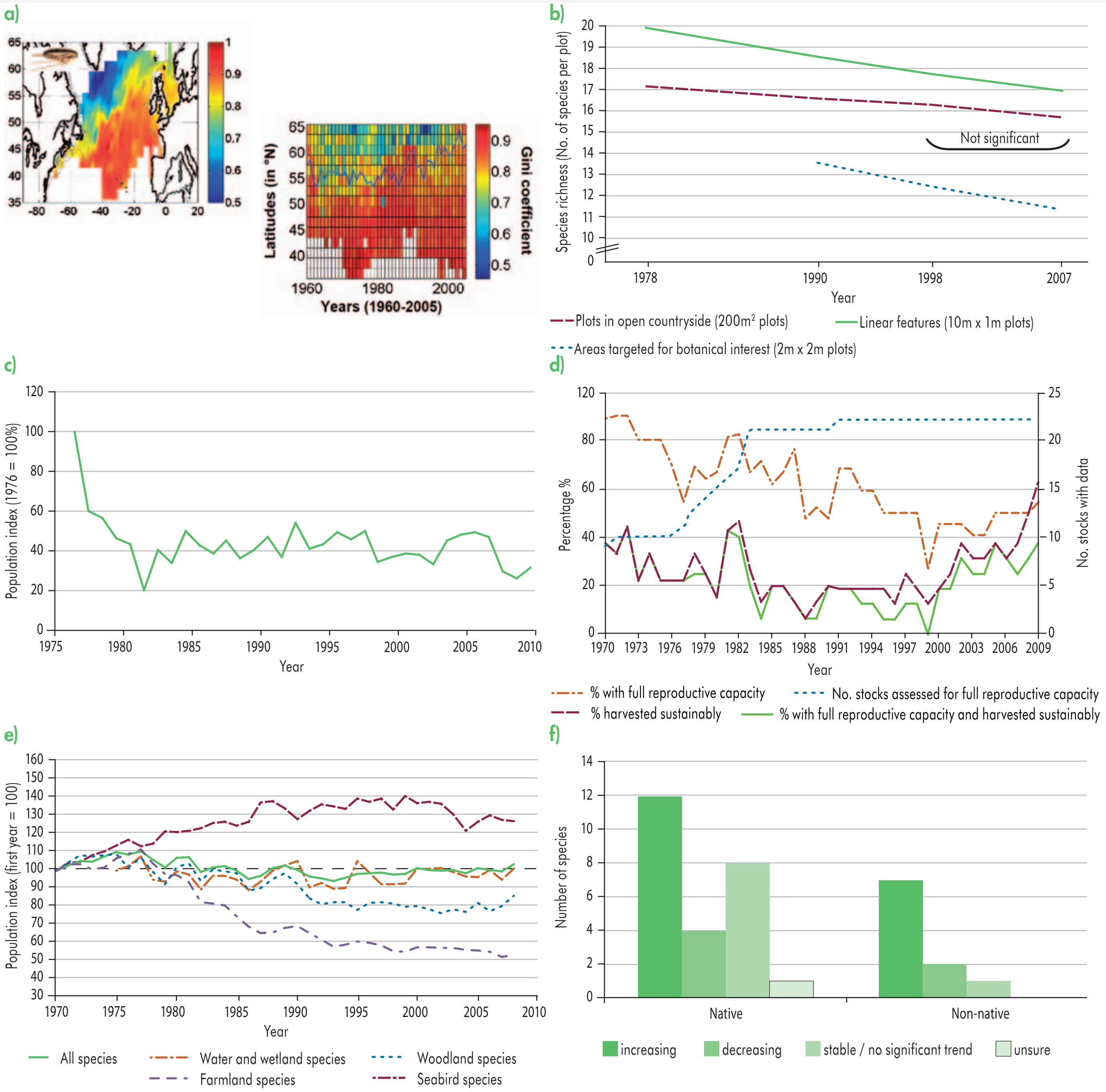


Figure 4.5 Status and trend information for selected UK biodiversity groups: a) The diversity of marine zooplankton communities. Diversity increases as colours move from blue to red. The left hand figure shows zooplankton diversity in the North Atlantic. The right hand figure shows that higher diversity regions have moved northwards over time. Source: reproduced from Beaugrand *et al.* (2010); **b) Average species richness of vegetation in plots in the open countryside (fields, woods, heaths and moors), linear features, and areas targeted for their botanical interest in GB between 1978 and 2007.** A decline in species richness is apparent in each dataset. Source: reproduced from Carey *et al.* (2008). Countryside Survey data owned by NERC – Centre for Ecology & Hydrology; **c) Composite population trend from 1976 to 2009 for 25 species of butterfly which are specialists of semi-natural habitats.** This demonstrates that populations have more than halved over the time period. Source: data from Butterfly Conservation, Centre for Ecology and Hydrology and Defra; JNCC (2010c);
d) The sustainability indicator for UK marine fin-fish stocks for 1990 to 2008 showing an improvement in sustainability from the late 1990s. Source: reproduced from Armstrong & Holmes (2010); **e) UK 'Quality of Life' indicators: Population trends of wild birds.** The graph shows the composite population trends of UK breeding bird species (n=114) with subdivisions showing grouped species' trends for seabirds (n=19), water and wetland birds (n=26), woodland birds (n=38), and farmland birds (n=19). On average, populations of woodland and farmland birds have fallen between 1970 and 2008 by 14% and 47% respectively. Source: data from RSPB, British Trust for Ornithology, JNCC and Defra; **f) Population trends in UK wild terrestrial mammal species up to 2007.** Sufficient data were available to assess population change for 35 species (n=25 native wild species, n=10 non-native wild species; this represents 53% of all UK terrestrial mammals). The data on the 11 species of native bat included in this summary are for 10 years to 2007; for all other species, trends were assessed over 25 years. Source: data from JNCC (2007).

limited evidence of population recovery in some species of conservation concern. For many other biodiversity groups, particularly some invertebrates, lower plants, fungi and microorganisms, data on status and trends are available for only a few localities and for comparatively short time periods (less than 10 years) in most cases. For these groups, recent status and trends are unclear. Details for each biodiversity group are given in **Appendix 4.1.**

4.5.3 Linking Status and Trend Information to Ecosystem Services

Assessing the impact of trends on ecosystem services, even for cases in which high quality trend information is available, is hampered by a lack of data on associated values and benefits. We illustrate this problem with the role of wild birds in cultural services. About 250 bird species regularly occur in the UK, and we have 40–50 years' worth of data on the distribution and abundance of the majority of these species (**Appendix 4.1**). In broad terms, seabird populations have increased, but have recently begun to decline; water and wetland populations have remained roughly stable, while woodland and farmland populations have declined (the latter to a greater extent); urban species have increased and wintering wader and wildfowl species have shown significant increases, followed by recent declines (**Table 4.3**). Conservation management has improved the status of a number of threatened species over the last 20 years (**Appendix 4.1**). Our assessment of biodiversity and UK ecosystem services suggests that birds play an important role in underpinning cultural services ('meaningful places' and 'socially valued landscapes and waterscapes') in the UK (**Table 4.2**). The impact of the population trends on these ecosystem services is unclear. Population declines in some habitats (e.g. farmland and woodland) might be expected to reduce service delivery (e.g. decreasing the value of socially valued landscapes and waterscapes); whereas increases in urban bird populations and charismatic species of conservation concern might be expected to have the opposite effect (e.g. increasing the value of meaningful places such as gardens or nature reserves). A lack of data on the various values and benefits people derive from wild birds associated with these cultural services, however, makes it impossible to quantify and integrate these potentially opposing effects in order to understand the net impact on each ecosystem service. This illustrates a critically important issue—even for biodiversity groups for which we have comprehensive data on status and trends, a lack of data on associated values and benefits often precludes a quantitative assessment of the impact of biodiversity changes on ecosystem services.

4.6 Drivers of Change

The drivers of change associated with each biodiversity group in the UK are detailed in **Table 4.5**. The drivers we distinguish here broadly follow those used in the MA, with one exception: since most habitat change in the UK occurs because of changes in land use and management, we have used the term 'land use change' to identify habitat changes arising from the way land is used and managed.

The trend information available for each biodiversity group was assessed as 'good' (UK-wide data on distribution, abundance and population trends over a 20-year or more time period), 'moderate' (UK-wide data on distribution, but limited data on abundance and population trends due to spatial or temporal coverage), 'patchy' (only localised data available on distribution or trends) or 'poor' (negligible data available on distribution or trends).

A number of points emerge from this overview:

- Land use change and pollution are considered the major drivers of change across biodiversity groups.
- Exploitation has a significant impact in marine ecosystems, both on target species, but also on non-target species through wider ecosystem changes.
- There is emerging evidence of climate change impacts across most biodiversity groups.
- The impact of invasive species on native biodiversity is considered less important for the majority of biodiversity groups, although there is evidence of impacts across a range of groups.

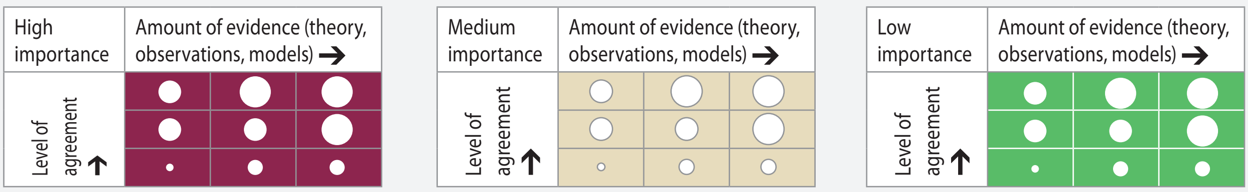
A comprehensive review of the evidence relating to the drivers of biodiversity change in the UK is beyond the scope of this chapter, but some general points can be made. Land use change is consistently assessed as an important driver of change across a wide range of biodiversity groups based on a large body of consistent, high quality evidence. For example, recent evidence suggests that two-thirds of the populations of 333 farmland species (broad-leaved plants, butterflies, bumblebees, birds and mammals) were threatened by agricultural intensification at the end of the 20th Century (Butler *et al.* 2009). Species with specialist ecological requirements (e.g. food types, nest sites) are more likely to decline in the face of land use change than generalists (Butler *et al.* 2007). There is evidence that land use change has reduced habitat heterogeneity in landscapes (Benton *et al.* 2003), thereby favouring generalist species that are able to reproduce and survive even in simplified landscapes (Smart *et al.* 2006). Biodiversity groups affected by land use change include those associated with estuarine, coastal and marine ecosystems due to the export of sediment and nutrients from land through aquatic ecosystems. Pollution impacts reflect a range of human activities including diffuse pollution from agriculture, and point source pollution from urban ecosystems and air pollution (e.g. acid rain); again, there is a large body of consistent evidence linking these activities with biodiversity changes (Bobbink *et al.* 2010; Maskell *et al.* 2010; Stevens *et al.* 2010). There is also evidence of some recovery from past large-scale pollution issues (Monteith *et al.* 2005). The impact of exploitation on target organisms, particularly in coastal and marine ecosystems, is well-documented (Cook *et al.* 1997). Exploitation also affects non-target organisms through physical and ecological changes to ecosystems (Votier *et al.* 2004).

There is emerging evidence of climate change impacts across a wide range of UK biodiversity groups (**Table 4.5**). The most compelling evidence comes from a northward shift in geographical range margins that have been described in terrestrial (Thomas & Lennon 1999) and marine ecosystems (Beaugrand *et al.* 2009), and from changes in the timing of important ecological events such as flowering in plants and breeding in animals (Thackeray *et al.* 2010). Non-native or invasive species represent a significant and increasing component of UK biodiversity (**Figure 4.5**). For example, recent evidence suggests that 117 non-native freshwater species are established, accounting for 12% of plant, 24% of fish and 54% of amphibian species richness (Keller *et al.* 2009). There is evidence that such species can have significant detrimental impacts on native biodiversity; well-documented examples include the decline of the native white-clawed crayfish (*Austropotamobius pallipes*) (Freeman *et al.* 2010) and water vole (*Arvicola terrestris*) (Rushton *et al.* 2000). There is also growing interest in the impact of non-native species on UK ecosystems (Lecerf *et al.* 2007). Taken together, this evidence suggests that climate change and invasive species are significant current drivers of biodiversity change in the UK, but, to date, these drivers have had a more limited impact than land use change, pollution and exploitation.

In response to these drivers of biodiversity change, there have been a wide range of changes in policy and practice designed to reduce biodiversity losses. There are examples of success in this respect: regulations to control pollution have led to improvements in water quality and the recovery of biodiversity in freshwater ecosystems (Monteith *et al.* 2005); management measures for marine fisheries have resulted in the recovery of some (10–20%) fish populations (**Figure 4.5d**); conservation legislation has promoted recovery among species of conservation concern (Donald *et al.* 2007); and recovery programmes for individual species of conservation concern have been successful (**Appendix 4.1**).

Table 4.5 Drivers of biodiversity change in the UK. This table is a synthesis from the accounts for different biodiversity groups (Appendix 4.1). Importance is colour-coded: high (maroon), medium (beige), low (green), unimportant on the basis of available evidence (blank). The size of the circle in each cell indicates the level of uncertainty. The impact of exploitation includes both the impact of the exploitation itself, but also the indirect consequences of exploitation through physical or ecological changes to the ecosystem.

Biodiversity Group		Trend information	Drivers of biodiversity change				
			Land use change	Climate change	Invasive species	Exploitation (direct and indirect)	Pollutants
Microorganisms	Marine	Patchy	Medium	Medium			High
	Terrestrial	Poor	High	Medium			Medium
Fungi	Non-lichenised	Poor	High	Low	Low		High
	Lichens	Moderate	High	Medium	Medium		High
Lower plants	Phytoplankton	Good		High	Medium		Medium
	Macroalgae	Patchy	Medium	Low	Low		High
	Bryophytes	Moderate	High	Low	Medium	Medium	High
Higher plants	Seagrasses	Patchy	Low	Low	Medium	Low	High
	Land plants	Good	High	Low	Medium	Medium	High
Invertebrates	Marine	Patchy	Medium	Medium	Low	High	Low
	Terrestrial	Moderate	High	Medium			High
Fish	Marine	Moderate		Medium		High	
	Freshwater	Good	High	Medium	Low	Low	High
Amphibians		Moderate	High	High	Medium		High
Reptiles		Patchy	High	Medium			
Birds		Good	High	Medium		High	Low
Mammals		Moderate	High	Low	Medium	Medium	



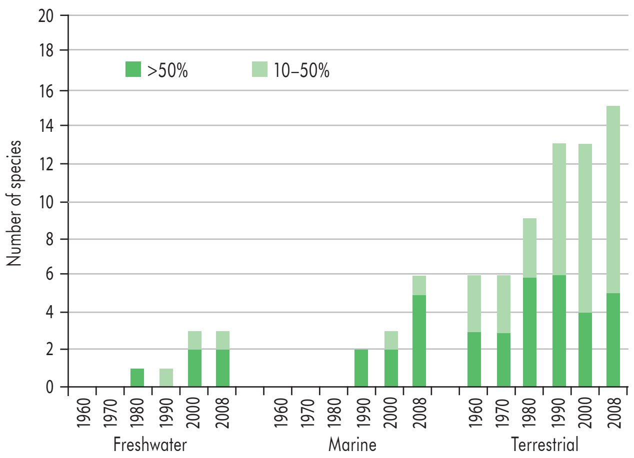


Figure 4.6 Changes in the extent of widely established invasive non-native species in freshwater, marine and terrestrial environments in Great Britain from 1960 to 2008. Source: data from the Centre for Ecology and Hydrology, British Trust for Ornithology, Marine Biological Association and the National Biodiversity Network Gateway; JNCC (2010d).

Despite these successes, significant biodiversity loss in the UK continues, mirroring recently reported global trends (Butchart *et al.* 2010). Land use change continues to drive the loss of terrestrial biodiversity in the UK despite the significant investment of public funds in schemes, such as agri-environmental management, designed to halt and reverse biodiversity losses. This is occurring because the adverse impacts of land use change on biodiversity have not been adequately removed by policy and practice measures (Butler *et al.* 2007; Butler *et al.* 2009). Addressing these issues requires improved spatial planning mechanisms to ensure that management measures appropriately target the adverse biodiversity impacts of land use change. Other drivers of biodiversity change, such as airborne nitrogen pollution, are difficult to address (Maskell *et al.* 2010). Finally, our understanding of the broad biodiversity impacts of drivers such as climate change and non-native invasive species remains limited, suggesting that further research will be needed on these issues. This need is pressing given that the impacts of climate change and invasive species are likely to increase (**Figure 4.6**). Taken together, this evidence suggests that where the impacts of drivers on biodiversity are relatively well-understood (land use change, pollution and exploitation), the effectiveness of policy and practice responses needs to be improved; whereas for other drivers (climate change and non-native invasive species), our understanding of biodiversity impacts needs to be improved before we can put into place effective responses.

4.7 Conclusions

While there is a clear evidence that biodiversity plays an important functional role in ecosystems and the services they deliver (Balmford & Bond 2005; Hooper *et al.* 2005; MA 2005), the available evidence that enables us to assess the importance of the role of different biodiversity groups in the context of ecosystem services in the UK is less than ideal. As a result, our assessment has been qualitative (low, medium or high) rather than quantitative. Nevertheless, a number of important findings have emerged. It is clear that UK ecosystem services have many important dependencies on biodiversity: all UK ecosystem services are dependent on biodiversity to some extent; more than 60% of the biodiversity groups assessed play an important role in underpinning the crops, plants, livestock and fish upon which we depend for food; and biodiversity is very important in a cultural context (**Table 4.2**). In addition, the finding that all UK ecosystem services are sensitive to changes in more than one biodiversity group is important because it suggests that UK ecosystems will need to support biodiversity across a wide range of groups to ensure the provision of a range of ecosystem services.

The UK has perhaps the most comprehensive data on biodiversity status and trends of any country in the world. These data clearly show that there have been significant range contractions and population declines over the last 40 to 50 years in a number of plant and animal groups; yet conservation efforts for some groups have improved the status of a number of threatened species in recent years (**Appendix 4.1**). Our assessment of drivers of biodiversity change suggests that land use change and pollution have played a major role in terrestrial and some marine ecosystems; whereas exploitation has been important for marine biodiversity groups (**Table 4.5**). There is emerging evidence of climate change impacts, and some evidence of detrimental effects of invasive species. All in all, this evidence shows that human activities in the UK have had significant, detrimental impacts on biodiversity, with important implications for ecosystem services given their dependencies on biodiversity.

Linking biodiversity change with changes in UK ecosystem services is, however, problematic because of the existence of a 'cultural divide' in our knowledge and understanding (**Figure 4.7**). On one side of the divide are biodiversity groups that underpin cultural services. We have shown that the quality of monitoring data on status and trends for a particular biodiversity group is related to its cultural importance (**Figure 4.4**). This means that we have high quality data on status and trends for culturally important biodiversity groups. However, we lack data on changes in cultural services associated with biodiversity change, making it extremely difficult to link the status and trend information to cultural services. We illustrate this problem with a case study on wild bird population trends (Section 4.5.3). On the other side of the cultural divide are provisioning and regulating services. We have high quality data on status and trends for these services (Chapter 14; Chapter 15), but frequently lack high quality monitoring data for key biodiversity groups—microorganisms, fungi and some plants—that underpin these services (**Table 4.2; Table 4.5; Figure 4.4**). This means we have a very limited knowledge of how these important biodiversity groups are changing, what drivers of biodiversity change are involved, and how any changes might affect provisioning and regulating services. Bridging this cultural divide represents

Figure 4.7 The cultural divide in our knowledge of biodiversity and associated ecosystem services. Good quality data are available (✓) on biodiversity associated with cultural services, but limited information (✗) on cultural services. In contrast, good quality data are available (✓) on provisioning and regulating services, but limited information (✗) on associated biodiversity.
Photos: Mackerel by Cory Doctorow; Water and glass by Stuart Olver†; Wheat © Stocker1970, 2011‡; Cumbrian landscape by HW-Photography¶; Dorset landscape © David Crosbie, 2011‡; Sunset by Gary Tanner†; Microbes by Microbe World†; Phytoplankton by willapalens†; Fly agaric fungus by Dave W. Clarke§; Bluebell wood by Angus Kirk¶; Honeybee (*Apis mellifera*) © Mirek Srb, 2011‡; Golden plover (*Pluvialis apricaria*) ©Leksele, 2011‡; Bluebell wood by Angus Kirk¶. Source: * available under a Creative Commons (CC) Attribution license Attribution-ShareAlike license; † available under a CC Attribution-NonCommercial-ShareAlike license; ‡ used under license of Shutterstock.com; ¶ available under a CC Attribution-NonCommercial-NoDerivs license; § available under a CC Attribution-NonCommercial license.*

perhaps the most important research and policy challenge relating to biodiversity in the UK.

To address this challenge, we need a shift in emphasis towards a more functional understanding of biodiversity in ecosystem dynamics (Nicholson *et al.* 2009). We need to: 1) *improve our understanding of how different biodiversity groups underpin ecosystem services*; 2) *identify key indicator groups, changes in which have an important impact on ecosystem services*; and 3) *develop a comprehensive, integrated monitoring programme for biodiversity in the UK around these indicator groups*. In making this shift, we need to recognise that the type of functional links between biodiversity and ecosystem services might vary between types of ecosystem service (Diaz *et al.* 2007). This has important implications for the way biodiversity is defined. Biodiversity groups that are currently recognised in a UK context largely reflect monitoring programmes developed around individual species within particular taxonomic groups. Improving our functional understanding of UK biodiversity will require a shift towards a more functionally relevant definition of biodiversity (Feest *et al.* 2009.; Schleuter *et al.* 2010; Vandewalle *et al* 2010.; Diaz *et al.* 2007). Despite the inevitable challenges involved, by adopting a more functional perspective we will be in a much stronger position to understand how biodiversity change in the UK impacts upon our ecosystem services in the face of future environmental and social change.

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Appendix 4.1

This section contains accounts for each biodiversity group written by experts from the UK's scientific community. Each account consists of: a broad definition of the group; a brief description of diversity in the UK; an outline of the group's role in ecosystem services; an overview of the available status and trends information; a description of the important drivers of biodiversity change; a view on future prospects; and a short list of key reference material.

A.4.1.1 Microorganisms

Authors: Mark Bailey, Sarah Turner, Paul Somerfield and Jack Gilbert

Taxa included in this group: Microorganisms range in size from 1–500 μm. Molecular systematics has revealed three major domains, the Bacteria and Archaea (Woese *et al.* 1990), formerly grouped as the prokaryotes, and the single-celled Eukaryotes. As a group they are genetically more diverse than all meso- and macro-fauna.

Diversity in the UK: Bacteria are the best described group of microorganisms. They have existed for around 3.5 billion years and represent the most diverse domain of life on Earth. Our understanding of bacterial biodiversity has been revolutionised by the use of molecular tools. More than 70% of bacterial taxa are known only from their DNA sequences and include taxa that are able to live in every known habitat on Earth, including plants and animals, soil, surface and subsurface water, deep in subsurface rocks and under extreme conditions of pH and temperature. The extent of microbial diversity has still to be fully described and, as such, we have little or no understanding of biogeographic or temporal trends either globally or at the UK-scale.

Roles in ecosystem services: Microbes constitute a major portion of the biodiversity and biomass in soils and water and, as a consequence, play an essential role in maintaining soil processes which ultimately regulate the functioning of ecosystems and the biogeochemical cycling of greenhouse gases (e.g. carbon in peat bogs). They are crucial to life, are central to all biogeochemical processes, and exist in spectacular numbers (Torsvik *et al.* 2002; Curtis & Sloan 2005). Yet the precise roles of the majority of microorganisms in ecosystem service delivery remain largely unknown (Raes & Bork 2008; Bell *et al.* 2005; Bardgett *et al.* 2005). It is well-documented that microorganisms recycle organic matter and minerals (carbon, nitrogen, sulphur, phosphorus etc.), purify water, biodegrade pollutants and colonise or infect plants and animals affecting their health status and susceptibility (DeLong 2009; Van der Heijden *et al.* 1998; Fierer & Jackson 2006). Only microbes possess the genes to fix essential nitrogen, and only microbes produce and oxidise biological methane (Battistuzzi *et al.* 2004). Microbial pathogens pose a direct threat to the health of humans and their domesticated animals and crops and hence directly affect food security. Less obviously, pathogens are important drivers of wildlife population dynamics and, as such, influence the net biodiversity of the UK. Microorganisms also provide vital services for food processing/production, for example, breaking down cellulose in the guts of ungulates, being of use in dairy products such as yoghurt and cheese, providing yeasts for the baking and brewing industries, etc.

Status and trends: Over the past three decades, our understanding of microbial biodiversity has been revolutionised by the use of molecular tools to characterise uncultivable communities and sequence entire genomes. For example, studies of the phylogeny of ribosomal gene and internal transcribed sequences (ITS), and other genes, have refined the Tree of Life, providing better insight into evolution and adaptation. Despite these advances, few data are available to determine what threats, if any, are posed to microbial populations.

Drivers of change: At present, we probably know too little to accurately relate microbial community structure to large-scale drivers (Raes & Bork 2008; Fierer & Jackson 2006; Fuhrman *et al.* 2006; Fuhrman 2009). Much is being revealed about the structure of communities (at a high taxonomic level) found in different functional habitats such as polluted streams, agricultural soils, marine waters, etc. (Heemsbergen *et al.* 2004; Pommier *et al.* 2007). Reliable data are emerging, for example, on the impacts of land use change and fertiliser inputs on soil microbial diversity (Bell *et al.* 2005; Fierer & Jackson 2006), or the influence of acidification and temperature on marine microbiota (DeLong 2009; Fuhrman 2009). But we know little of the impacts of these drivers on function and sustainability, or how the structure and functioning of communities is affected by climate or environmental change.

Prospects: This is an area where technological advances are increasing knowledge on a daily basis, and where step changes in understanding ecosystem processes are coming of age. To date, much of the research into microbial ecology has focused on describing diversity and identifying patterns in relation to environmental parameters. More recently, however, the integration of high through-put sequencing approaches with *in situ* process measures is revealing how microbial community structure affects or responds to aspects of human, animal and plant health (including susceptibility to disease), environmental quality, nutrient cycling and status. Because of their short lifecycle, microbes have the potential to act as indicators of immediate and short-term change, and should lead us towards new approaches to manage health and environmental risks.

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A.4.1.2 Fungi and Lichens

Non-lichenised fungi

Author: Stephan Helfer

Taxa included in this group: This group includes fungi with the exception of lichenised species, which are dealt with separately below. Although they are parts of other lineages, for this treatment 'fungi' includes Mycetozoan (e.g. Myxomycota) and Heterokontophytan (e.g. Oomycota) species. Fungi are eukaryotic, heterotrophic organisms, and obtain their nutrients by absorption. They reproduce by sexual or asexual spores. Most fungi have a thallus composed of hyphae that elongate by tip growth.

Diversity in the UK: There are approximately 12,000 species of fungi known in Britain (Hawksworth 1991), which represents approximately 1% of global, and possibly 20–30% of European, diversity. They range from single-cell yeasts or purely soil-inhabiting asexual organisms, such as the Glomeromycota, to mutualistic, saprobic or parasitic macromycetes with prominent fruiting bodies, commonly known as 'mushrooms'. Many species are in mycorrhizal associations with higher plants; other species are obligate or necrotrophic plant or animal parasites, or litter decomposers. While some of the fungi, notably sand dune fungi, waxcaps and tooth fungi, have received considerable attention from conservation biologists, the majority of fungi have not been assessed. Conservation for sand dune fungi, waxcaps and tooth fungi is well-established. However, while it has been on the agenda for some time, conservation for fungi in general has only been pursued systematically since the 1990s in GB. Currently, there are 76 priority species of non-lichenised fungi in the JNCC list of UK Biodiversity Action Plan's priority species, 10 of which are micromycetes (Anon 2007).

Roles in ecosystem services: Many crop plants have mycorrhizal associations with fungi (Sawers *et al.* 2008) and benefit greatly from this arrangement with an estimated 100–900% biomass added compared with non-mycorrhizal controls in nutrient-poor soils. This is equally the case for wild plants and some groups of invertebrates (Bärlocher 1985). Conversely, parasitic fungi cause serious losses in crops (around 10–15% pre-harvest loss) and wild plants (e.g. *Phytophthora* species), and can have devastating effects on wild fauna (e.g. chytrids attacking amphibians). Forest and amenity trees are equally impacted by both sides of the fungal spectrum (e.g. ectomycorrhizal fungi versus Dutch Elm Disease). Common wood-rotting fungi produce high levels of chlorinated aromatics, having a direct impact on pollution and climate regulation (Jong *et al.* 1994). Furthermore, in conjunction with methanogens, fungi in the guts of ruminants are responsible for most of the biogenic production of greenhouse gases (Moss *et al.* 2000). The mycelium of soil fungi is responsible for soil water absorption and retention capacity (Rillig *et al.* 2010). However, overall effects on water quantity are poorly understood. Being capable of degrading lignin, fungi are responsible for the majority of plant litter breakdown and nutrient recycling (Steffen *et al.* 2007). They also assist soil invertebrates in the digestion of plant litter (Douglas 2009). Moreover, fungi are used in the detoxification of oil spills, spoil hills (bings) and even radioactive and other hazardous waste.

Status and trends: While many new species have been added to the UK fungi Red Data List since 1992, and many have increased their rating of vulnerability (69 more extinctions and 21 more Critically Endangered; e.g. *Puccinia scirpi*), some species have had their threat status lessened (7 species no longer Extinct and 24 less Critically Endangered; e.g. *Xeromphalina picta*) (Evans 2007; **Figure A.4.1.2.1**). With this in mind, a general trend for fungi is currently impossible to establish. This is because most, if not all data are deficient, and many fungi 'turn up' or 'disappear' unexpectedly, whereas the mycelium is present all the time.

Drivers of change: The main threats to fungi are pollution, and agricultural and forestry management practices, in particular, nitrogen input and pesticide use, but also the choice of tree species in forest plantations. Land use changes, especially road-building and the development of housing, also threaten species. Conversely, some of these threats have increased the opportunities for fungi.

Prospects: There is little evidence to suggest an overall threat to fungal biodiversity in GB. As with lichens (see A.4.1.2), generally, fungi show a loss of 'northern' species in southern GB and a gain in 'southern' species, coupled with a change in fruiting phenology (Kauserud *et al.* 2010).

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Figure A.4.1.2.1 a) The number of threatened fungi in the UK, and b) threat categories, in 1992 and 2007 respectively. The total number of threatened species has increased, as have the threat categories. This is mainly attributable to greater data availability and does not necessarily indicate an increase in threat. Source: data from Evans (2007).

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Lichens

Author: Christopher J. Ellis

Taxa included in this group: This group includes the lichenised-fungi. Lichens represent a symbiotic relationship between a heterotrophic fungus and a photosynthetic alga or cyanobacteria, which occurs internally within a fungal-derived macrostructure, the lichen thallus. Approximately 98% of lichenised-fungi are Ascomycetes ('cup'-fungi), with the majority of the remaining 2% being Basidiomycetes. The lichenised-fungi are, therefore, a functional group, and are not monophlyletic; within Ascomycetes, lichens are formed by fungi occurring in several major clades.

Diversity in the UK: There are around 1,900 species of lichenised-fungi (lichens) in the GB, and a further 428 species of lichenicolous fungi (though this number is increasing). This represents about 47% of European lichen diversity. Of the 2,331 taxa representing lichens and lichenicolous fungi, 17% warrant an International Union for Conservation of Nature (IUCN) threat category, with a further 1.3% considered extinct and 11% considered data deficient (Woods & Coppins 2010). GB has 'international responsibility' for 185 species (representing 8% of the flora). This accounts for lichens with viable populations in GB, but which are extremely rare or threatened across Europe or the world, e.g. *Lobaria pulmonaria*.

Roles in ecosystem services: Evidence for the role of lichens in contributing to the diversity of wild species is extremely high (see Diversity in the UK above). *En masse*, lichens also significantly contribute to the aesthetic character of celebrated landscapes in GB, though individual lichens may themselves go unnoticed; for example, the lichen-rich tundra vegetation of the Cairngorm plateau provides an Arctic-feel to the landscape, while epiphyte flora defines the character and sense of place of the 'Celtic Rainforest'. Within these and other ecosystems, lichens may provide an essential ecosystem role, for example, cyanobacterial photobionts fix nitrogen from the atmosphere, contributing to nutrient cycling (Antoine 2004), and there is a functional relationship between lichen-epiphyte biomass and the diversity of food-source invertebrates for birds (Pettersson *et al.* 1995; Gunnarsson *et al.* 2004). This role may extend to hazard regulation: lichens as primary colonisers may play a role in soil stabilisation and vegetation succession, for instance.

Status and trends: For a majority of lichen species there are insufficient data to provide an empirical account of population and metapopulation change. However, a number of general trends are apparent, or have been inferred, indicating a response to both large-scale environmental drivers and local habitat.

Lichen diversity is recovering in regions where biodiversity was decimated during a period of severe industrial pollution from the 19th Century onwards (Coppins *et al.* 2001). This process of recovery is attributed to reduced sulphur dioxide pollution and acid rain, though is curtailed by the concurrent increase in environmental nitrogen inputs, which appear to prevent the full recovery of the lichen flora and may, in fact, have additional and severe negative impacts (van Herk *et al.* 2003; Wolseley *et al.* 2006; Ellis & Coppins 2009). Sensitivity to nitrogen pollution—most commonly examined for epiphytes—extends to include important areas of upland lichen-rich heath (Britton & Fisher 2007), and is thought to be a key factor driving a homogenisation of montane vegetation (Britton *et al.* 2009).

Lichens are sensitive to habitat dynamics, and many species are habitat specialists, growing on particular rock types or as epiphytes on certain tree species, or growing in local climatic settings or on substrata of a particular age. This specialisation makes lichens extremely vulnerable to habitat loss or degradation. Additionally, recent research indicates an 'extinction debt' for lichens (Berglund & Jonsson 2005; Ellis & Coppins 2007): a process by which populations of patch-tracking and long-lived lichens may exist within remnant habitat patches (following the loss of adjacent habitat), but where subsequently declining populations may not be replaced by recolonisation following a process of landscape change. Owing to this delay, extant lichen diversity patterns may not reflect current landscape structure, but may be related to patterns of historic habitat quality. This process indicates a far wider implication of habitat change for lichens (perceived across landscapes), outside the immediate loss and degradation of local habitat.

Lichens are expected to be sensitive to climate change (Ellis *et al.* 2007); there is preliminary observational evidence for declining populations of 'northern' species in southern GB, and newly discovered populations of 'southern' species in northern GB (Ellis & Binder 2007. This preliminary evidence for GB matches with more robust, long-term evidence for continental Europe, which has demonstrated an increase in warm-temperate or subtropical species, and a loss or northward shift in the distribution of boreal species (van Herk *et al.* 2002; Lättman *et al.* 2009).

Drivers of change: Lichens are sensitive to large-scale environmental drivers, such as pollution and climatic setting (including climate change), and, as diminutive organisms nested within a larger-scale habitat matrix, lichens are sensitive to changes in land management impacting local habitat dynamics.

Prospects: The prospects for lichens are uncertain: assuming pollution (a blanket form of 'habitat loss') is reduced, the persistence of lichen diversity is dependent on maintaining a sufficiently high density of high quality mixed habitats within the landscape. This would allow species richness to be maintained through colonisation-extinction dynamics, despite species compositional turnover in response to climate change. Considering the potential importance of habitat quality in buffering an amalgam of human impacts (pollution, climate change), it is of critical importance that 45% of conservation sites, specifically Sites of Scientific Interest (SSSIs), notified for lower plants (including lichens) are in unfavourable condition.

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A.4.1.3 Lower Plants

Phytoplankton

Author: Martin Edwards

Taxa included in this group: Phytoplankton include all photoautotrophic microorganisms found in aquatic ecosystems, e.g. diatoms, cyanobacteria, dinoflagellates and coccolithophores. Collectively, they inhabit more than 70% of the Earth's planetary surface.

Diversity in the UK: The diversity of phytoplankton is not truly quantified in the UK; however, long-term studies suggest an increase in phytoplankton diversity associated with increasing temperature. Phytoplankton diversity around the British Isles is similar to other temperate/boreal marine ecosystems. Diatom diversity is highest in the southern North Sea and other regional seas on the Northern European continental shelf associated with mixed-water habitats. Overall, phytoplankton diversity (including dinoflagellates and coccolithophores) is highest in oceanic waters to the west of the GB where seasonal stability and temperatures are higher.

Roles in ecosystem services: Phytoplankton have roles in: global climate regulation, global oxygen production, global carbon sequestration, nutrient recycling and primary energy transformation; and are the foundation of virtually all marine food chains, leading to global fish production and other marine bioresources.

Status and trends: At the ocean basin-scale, studies on pelagic biodiversity have been related to temperature—an increase in warming over the last few decades has been followed by an increase in diversity. In particular, increases in diversity are seen when previously low diversity systems, such as Arctic and cold-boreal provinces, undergo prolonged warming events. The overall diversity patterns of pelagic organisms, peaking between 20° to 30° north or south of the equator, follow temperature gradients in the world's oceans. Phytoplankton show a relationship between temperature and diversity which is linked to the phytoplankton community having a higher diversity, but an overall smaller size-fraction and a more complex food web structure (i.e. microbial-based versus diatom-based production) in warmer, more stratified environments. Climate warming will, therefore, increase planktonic diversity throughout the cooler regions of the world's oceans as temperature isotherms shift poleward. Long-term phytoplankton data, collected by the Continuous Plankton Recorder since the 1950s, has shown an increase in phytoplankton diversity around the British Isles associated with sea surface warming and climate oscillations.

It has recently been highlighted that Arctic ice is reducing faster than previous modelled estimates. As a consequence, the biological boundaries between the North Atlantic Ocean and Pacific may become increasingly blurred, with an increase of trans-Arctic migrations becoming a reality. The Continuous Plankton Recorder survey has already documented the presence of a Pacific diatom, *Neodenticula seminae*, in the Labrador Sea, which was first observed in the late 1990s and has since spread southwards and eastwards. The diatom species itself has been absent from the North Atlantic for more than 800,000 years and could be the first evidence of a trans-Arctic migration in modern times, as well as the harbinger of a potential inundation of new organisms to the North Atlantic. The consequences of such a change to the function and biodiversity of Arctic systems are, at present, unknown.

Drivers of change: Generally, the changes in phytoplankton biodiversity at the oceanic macroscale appear to be mainly driven by changes in temperature. There is a strong relationship between temperature and pelagic biodiversity with higher temperatures leading to higher biodiversity. Seasonal stability of the water column has also been shown to increase phytoplankton diversity. During the last decade, in open ocean systems around the British Isles, planktonic biodiversity has increased in association with higher sea surface temperatures. There is some evidence that localised nutrient enrichment caused by terrestrial runoff can cause mono-specific phytoplankton blooms in coastal regions (i.e. a decrease in diversity); however, these blooms have tended to be temporally transient events. Ocean acidification may have future consequences for some calcifying phytoplankton such as coccolithophores.

Prospects: In oceanic habitats free from coastal anthropogenic influences (assuming no habitat loss/fragmentation in oceanic systems), it is highly likely that phytoplankton diversity will continue to increase in association with higher temperature projections. Although the traditional anthropocentric viewpoint considers increasing biodiversity as a positive attribute for ecosystems (e.g. increased homeostatic stabilising processes, decreased energy loss from systems, move towards closed nutrient recycling, etc.), increasing phytoplankton diversity may have a number of negative consequences. For example, the increase in phytoplankton diversity is strongly associated with a decrease in the size-structure of the community, leading to the energetic dominance of smaller organisms. In turn, this has consequences for both carbon residence times (increase in carbon residence times in surface waters) and the size-structure of other ectotherms such as fish. Increasing phytoplankton may, therefore, lead to the devaluation of fisheries, with a move towards smaller species and communities. Increasing temperatures and diversity may also lead to floristic shifts from diatoms to flagellates, potentially leading to more occurrences of harmful algal blooms.

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Macroalgae

Authors: Olivia Langmead and Emma Jackson

Taxa included in this group: The taxa in this group include all multicellular eukaryotic algae belonging to one of three main groups: red algae (Rhodophyta), green algae (Chlorophyta) and brown algae (Phaeophyceae).

Diversity in the UK: Macroalgae are a polyphyletic group, which is not only taxonomically diverse, but also displays high functional diversity. There are approximately 800 nationally recorded species of algae in the UK, from about 100 families, 7 phyla and 2 kingdoms (Plantae and Chromista). A recent assessment of global diversity of macroalgae identified UK waters as part of a cluster of endemics in Western Europe (Kerswell 2006).

Roles in ecosystem services: Macroalgal beds are important coastal habitats and support a variety of ecosystem services including providing feeding and nursery habitat for many commercially important fish and shellfish species, and opportunities for direct harvesting for food, biofuels and pharmaceutical products (and historically for fertiliser). Macroalgae also play regulatory roles in the transformation of nutrients to organic matter and their export to other systems, the regulation of oxygen, and the breakdown and removal of pollutants.

Status and trends: No UK-wide assessment of status and trends in marine macroalgae has been undertaken to date, and our perspective of this diverse and functionally important group is incomplete. Data is being collected from Special Areas of Conservation (SACs) in terms of site condition-monitoring for areas designated for rocky reefs, but no overview reports have been produced to summarise the trends or current status. The high functional diversity of this group makes it difficult to generalise about their status and trends. However, there is evidence that changes follow wider European trends where the loss of long-lived, slow-growing, functionally important macroalgae to more opportunistic species has been associated with anthropogenic pressure (Orfandis *et al.* 2003).

Drivers of change: Land use change and pollutants are key drivers of change in status of macroalage. Nutrient enrichment, typically caused by terrestrial runoff of treated sewage and agricultural fertiliser, can lead to changes in macroalgal communities, with outbreaks of opportunistic species, such as *Ulva* species, occurring (Fletcher 1996). These algae compete with other species for space, as well as generating large quantities of biomass which, upon decomposition, may locally reduce oxygen levels among intertidal habitats. In comparison, slow-growing, long-lived algae, such as maerl (calcified red seaweed), are damaged by dredging, anchoring and eutrophication. Between 1982 and 1992, the proportion of dead maerl on the St. Mawes bank, Cornwall, increased significantly from 12% to 23% (Perrins 1995); it also increased at Milford Haven, South Wales, due to industrial coastal development (Jackson *et al.* 2008).

Invasive macroalgal species (e.g. *Sargassum muticum, Undaria pinnatifida, Asparagopsis armata*) are a growing concern, but impacts on native communities have not been consistent (Milneur *et al.* 2008). Assessments of ports show many species are present, potentially acting as reservoirs for further spread in UK waters (Arenas *et al.* 2006). Research on the distribution and abundance of northern (cold water) macroalgal species has shown climate change related range contractions and significant declines in abundance (e.g. cold water brown macroalga, *Alaria esculenta*). This trend is accompanied by range extensions in warm water species (e.g. southern red turf alga, *Chondrocanthus acicularis*, warm water kelp, *Sacchoriza polyschides*) (UKMMAS 2010; Mieszkowska *et al.* 2006).

Prospects*:* Our understanding of the status and trends will improve with the implementation of monitoring for the Water Framework Directive. Macroalgae are a biological quality element to be used in defining the ecological status of transitional and coastal water bodies. Opportunistic macroalgae will be assessed in coastal and transitional waters (Scanlan *et al.* 2007), macroalgal community structure will be assessed on rocky shores (Wells *et al.* 2007, and fucoid extent will be assessed in estuaries (Wilkinson *et al.* 2007).

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Bryophytes

Authors: David G. Long, David F. Chamberlain and Elizabeth Kungu

Taxa included in this group: The group includes the liverworts (Marchantiophyta), mosses (Bryophyta) and hornworts (Anthocerotophyta), which are quite independent evolutionary lineages, but are placed together for convenience because they are similar in size, share alternation of generations with a dependent sporophyte and occupy very similar ecological and biological roles in nature. The liverworts include the earliest known lineages of land plants.

Diversity in the UK: There are 1,056 species of bryophytes in the GB: 297 liverworts, 755 mosses and 4 hornworts (Hill *et al.* 2008). This constitutes approximately 65% of European species and 6% of the estimated total global species. A recent assessment (JNCC 2007) categorises 22% of British bryophytes (231 species) as Red List species based on IUCN guidelines. Additionally, there are numerous species whose UK populations do not warrant formal conservation status, but are of high international importance, such as liverworts with disjunct global distributions found in rare and vulnerable habitats including Oceanic Liverwort Heath (Rothero 2003) and Atlantic Oakwoods (Hodgetts 1997).

Roles in ecosystem services: Bryophytes contribute significant diversity to almost all GB ecosystems. Where they dominate ecosystems, they are most visible and often highly aesthetically attractive, for example, the rich colours of the 35 British *Sphagnum* species in our northern peatlands, and the sheer luxuriance of mosses and liverworts clothing our oceanic woodlands. In other habitats, they are less conspicuous, but still vitally important, for example, as stabilising colonists of coastal sand dunes, or forming carpets of tiny Arctic species around Cairngorm snowbeds (our own 'tundra'). Peatlands (consisting of both living and decaying *Sphagnum*) are of exceptional importance for carbon sequestration: globally they contain 320 billion tonnes of carbon, about 44% of the amount held in the atmosphere as carbon dioxide (Rydin & Jeglum 2006). Peatlands are also a major source of fuel, energy and horticultural growing mediums (Vanderpoorten & Goffinet 2009). Bryophytes are drought-tolerant (poikilohydric) and have very high water retention properties—up to 1,500% of their dry weight (Proctor 2009)—which is significant in peatlands and mossy forests where high rainfall can be absorbed by the plants preventing rapid runoff and flooding, and maintaining humidity through dry seasons. In many communities, they act as pioneers and stabilise soil crusts. Physically, they provide microhabitats for many invertebrates, which, in turn, provide food for a variety of organisms, particularly in aquatic and mossy forest ecosystems (Lindo & Gonzalez 2010; Parker *et al.* 2000).

Status and trends: Progress in recent years has been considerable in mapping changes to bryophyte distributions in GB (Hill *et al.* 1991–1994), but at the population level, data are in most cases, inadequate to assess trends in any depth. A number of trends can be identified from anecdotal and other evidence, however:

Habitat loss, degeneration and fragmentation. Many bryophytes display highly specialised habitat requirements such as epiphytes showing sensitivity to bark pH, 'copper mosses' restricted to metalliferous rocks, woodland species dependent on shade and high humidity, rheophytes dependent on fast-moving water and montane species dependent on late snow-lie. Such demanding traits render the bryophytes highly vulnerable to habitat loss and degradation, particularly as many of these niches occur on a very small, local scale and may be highly isolated. However, many bryophytes have developed life strategies to deal with ecological instability, and those species with effective dispersal capacity may cope in the face of such changes; others, such as the oceanic-montane liverwort heath species, lack reproductive capacity and have no such defences against habitat change, particularly when it is rapid as a result of ecological calamities such as muirburn (Hobbs 1988).

*Pollution***.** Although primary consideration is often given to air pollution due to its historically destructive impact on bryophyte and lichen epiphytes, both water pollution and agricultural runoff on land can also impact negatively on the bryoflora (Vanderpoorten & Goffinet 2009). The effects of sulphur dioxide pollution have clearly been demonstrated by the historic losses of mosses, such as epiphytic *Orthotrichum* species, in industrial areas, and their recent return to many urban places. However, the effects of nitrogen oxide pollution may be increasing, even in montane areas remote from industrial activity (Woolgrove & Woodin 1996). This may lead to the loss of more demanding and sensitive species.

Climate change. Bryophytes are predicted to show high sensitivity to climate change, with evidence that some northern species are declining in southern GB, and some southern species, such as *Grimmia tergestina*, are moving northwards (Porley & Hodgetts 2005); those already close to their altitudinal limit, such as Arctic bryophytes of snow-beds, may have no way of escape. Not all changes may be temperature-driven, and there is already anecdotal evidence that, in the face of increasing summer rainfall, some oceanic liverworts, such as *Metzgeria temperata*, are moving eastwards. An even more dramatic consequence may be the effect of global warming on peatlands which has been described as a 'ticking time bomb' (Vanderpoorten

& Goffinet 2009) as their decomposition could increase atmospheric carbon dioxide by up to 50% (O'Neill 2000).

Drivers of change: Bryophytes show many similarities to lichens in their sensitivity to environmental drivers such as climatic effects and pollution. Where they occur as components of larger-scale ecosystems, such as in forests and heathlands, they are especially sensitive to human-induced habitat changes at a landscape-scale, such as agricultural intensification, burning, grazing and afforestation.

Prospects: Awareness of the importance of bryophytes in the face of climate and other environmental challenges will surely increase—it is already reflected by their collective description as the 'bryosphere' (Lindo & Gonzalez 2010). The prospects for bryophytes are uncertain. Even with possible reductions in pollution, the relentless pressure on bryophyte-rich habitats shows little sign of diminishing. Historically, our 'protected' areas have been selected to reflect more charismatic interests, and often the richest bryophyte sites are undesignated and under-managed. Even many Sites of Special Scientific Interest notified for cryptogamic plants are in unfavourable condition. A realignment of priorities to more equitably allocate resources on research and conservation to less-charismatic groups, such as cryptogams and invertebrates, is long overdue in GB.

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A.4.1.4 Higher Plants

Seagrass

Authors: Emma Jackson and Olivia Langmead

Taxa included in this group and diversity in the UK: There are two species of seagrass on the coasts of the UK, the primarily subtidal eelgrass (*Zostera marina*) and the intertidal dwarf eelgrass (*Zostera noltii*). Currently, there is debate as to the status of a possible third species, *Zostera marina var. angustifolia*, which is not widely accepted as a separate species. Dwarf eelgrass is at its most northerly biogeographical limit in the UK.

Roles in ecosystem services: In the UK, seagrass meadows function as important nursery and foraging habitat for fish, shellfish and wildfowl, and they also oxygenate and stabilise sediments and store carbon. They are considered a 'foundation species' i.e. organisms that provide habitat, enhance ecosystem biodiversity and are important indicators of system health.

Status and trends: There is no national monitoring programme for seagrasses in the UK and, therefore, overall trends are difficult to assess. Global trajectories show accelerating losses of seagrasses over the last 100 years (Waycott *et al.* 2009). In the UK, seagrass beds have never fully recovered from large-scale losses attributed to the 1930s outbreak of 'wasting disease' due to the significant changes in sediment dynamics it caused (Wilson 1949). During the 1990s, repeated outbreaks of wasting disease led to further seagrass losses in the Solent (Chesworth *et al.* 2008).

Information from the few UK monitoring studies that do exist demonstrate trends of loss that continued through the 1990s. However, this pattern varies spatially; sublittoral beds in Pembrokeshire have undergone recovery in some areas, such as North Haven, and entirely disappeared in others (Foden & Brazier 2007). Annual diving surveys of seagrass beds in the Isles of Scilly have shown inconsistent trends (in shoot density) between five meadows over the past 13 years suggesting localised effects on the beds rather than large-scale pressures (Cook 2005). The current spatial distribution of seagrass in the Solent appears to be consistent with records from the 1980s (Lefebvre *et al.* 2009).

Intertidal beds (dwarf eelgrass) show similar diverse trends around the UK; seagrass extent does not seem to have greatly changed over the last two decades within the Solent (Lefebvre *et al.* 2009). In Welsh coastal waters the number of intertidal seagrass beds has increased during this century (Boyes *et al.* 2009).

Drivers of change: Changes in seagrass health and distribution are driven primarily by land use change, indirect exploitation and invasive species. Activities which decrease water clarity or quality (for example, eutrophication, aquaculture, coastal development, dredging and spoil disposal) may negatively impact the health or productivity of seagrass (evident in density, biomass or epiphyte cover changes). Increased turbidity may also reduce the depth limit and thus vertical distribution of the seagrasses. While nutrient enrichment may increase production in seagrasses, associated phytoplankton blooms and opportunistic algal growth (including epiphytes and invasive species such as wireweed, *Sargassum muticum*) may cause severe shading. Boat anchoring, propeller scarring, dredging and destructive fishing methods, such as beam trawling, have all been shown to physically damage seagrasses. Damage to UK seagrass beds from such activities are evident in visual surveys (Rhodes *et al.* 2006; Sutton & Tompsett 2000).

Prospects: Improvements in water quality through improved sewerage treatment and national regulations resulting from the Urban Waste Water Treatment Directive and Water Framework Directive have started to negate pressures relating to water clarity and quality. However, continued direct physical pressures on seagrass beds are increasingly resulting in the loss and fragmentation of many beds. Increased storm events predicted as part of a changing climate are likely to have negative effects on the current extent of seagrass beds, making them more vulnerable to more direct human drivers of change. Increased availability of inorganic carbon due to ocean acidification is likely to have positive benefits for the growth and health of seagrasses, but may disrupt the ecosystem services they provide (Hall-Spencer *et al.* 2008).

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Land plants

Authors: Mary Gibby, Heather McHaffie and Lindsay Maskell

Taxa included in this group: This group includes all vascular plants: Lycopods, Isoetes and Selaginella, ferns and horsetails, conifers (Gymnosperms), and all flowering plants (Angiosperms)—trees, shrubs, herbaceous plants and grasses. The majority are land plants, but some occur in freshwater, brackish or marine habitats.

Diversity in the UK: With approximately 1,500 species, the vascular plant flora of the GB is considered depauperate in comparison with that of mainland Europe, constituting only 13.6% of European species and 0.47% of the estimated total global species. Most of the 1,500 native species are angiosperms; ferns, horsetails and Lycopods account for around 80 taxa, and there are just three conifers (Scots pine, juniper and yew). *The New Atlas of the British and Irish Flora* (Preston *et al.* 2002) includes 1,486 native and 817 introduced species; these latter are classified as 'archaeophytes' (introduced before AD 1500), 'neophytes' (introduced after AD 1500) and 'casuals' (recorded but not forming permanent populations).

Roles in ecosystem services: Vascular plants are significant components of ecosystems, delivering both provisioning and regulating services, with a major role in carbon cycling and oxygen release. Land plants are the basis of productive agriculture and forestry. Forests act as carbon sinks and influence climate change by affecting the amount of carbon dioxide in the atmosphere. Forest cover helps regulate rainfall, slows down runoff and reduces erosion. Wild species diversity is ultimately dependent on vascular plants; they form the base of the food chain and provide shelter and a diversity of habitats. Of the 65 UK Biodiversity Action Plan priority habitats, 33 are dominated by vascular plants, from various woodlands of upland and lowland, heath and scrubland, hay meadows and chalk grassland, to traditional orchards, hedgerows and coastal sand dunes. Vascular plants have a key role in water purification. They are the framework for meaningful places, promote health and well-being, and are valued in green landscapes.

Status and trends: The revised Red Data List (Cheffings *et al.* 2005) includes native species and archaeophytes: 125 species are Endangered or Critically Endangered, 220 are Vulnerable and 98 are Near Threatened. This means that 443 species are of conservation concern—some 25% of the native and archaeophyte flora. The Countryside Survey is a monitoring scheme which provides quantitative data on changes in plant species distribution across GB between 1978 and 2007 (www.countrysidesurvey.org.uk/). It samples a series of plots representing different landscape features within a 1 km square (591 x 1 km^2 plots were sampled across GB in 2007). Data from the Countryside Survey contributes to the UK Biodiversity Indicators (www.jncc.gov.uk/page-4229). Results from 2007 showed that there had been a decline in mean species richness in most habitats in GB between 1978 and 2007 (Carey *et al.* 2008a; **Figure A.4.1.4.1**).

Within the UK, countries have produced their own analysis of species and habitats giving added conservation status to local species as seen in the Scottish Biodiversity List (2005) and the Vascular Plant Red Data List for Wales (Dines 2008). The Countryside Survey also reports by country and there are summary reports for England, Wales and Scotland. Ireland's National Biodiversity Plan (2010) builds on the Global Strategy for Plant Conservation (2002).

A detailed study of local change was undertaken by members of the Botanical Society of the British Isles (BSBI) compiling datasets gathered in 1987–1988 and again in the same 2 x 2 km areas in 2003–2004 (Braithwaite *et al.* 2006), providing a baseline that can be further extended. Comparisons over a shorter period than between the two Atlases indicate the loss of species from less fertile habitats, calcareous grassland and dwarf shrub heath, probably due to habitat fragmentation. Climate change appears to be favouring the spread of some southern species, but there is less evidence of decline in the uplands as yet.

Drivers of change: There are many reasons for decline. The most outstanding is probably habitat loss, particularly in plants that have suffered from 'improvement' and drainage of wet areas. This is clearly illustrated in the distribution of pillwort (Pilularia globulifera), a small fern that depends on light disturbance to reduce competition and is sensitive to water quality. This species is declining throughout Europe, but the GB populations are comparatively strong which gives them an added importance (**Figure A.4.1.4.2**). The existing records are usually in less populated areas.

Other species have also suffered from more intensive agriculture, and the increased use of herbicide has reduced the abundance of many arable species, which are not nearly as frequent as they used to be, although they are still present in many areas. The archaeophyte, corn spurrey (*Spergula arvensis*), for example, has a change index of -2.30 and if the trend continues might disappear within the next 40 years. Conversely, the native lesser sea spurrey (*Spergularia marina*) has an expanded range inland on salted road margins and has a change index of +1.83.

As well as direct impacts on arable species, as mentioned above, the burning and processing of fossil fuels for energy production, and the manufacture and application of agricultural fertilisers, have resulted in large-scale increases in macro-nutrient inputs to ecosystems. Eutrophication signals have been detected across all habitat types (Smart *et al.* 2003). Eutrophication leads to loss of diversity, particularly when coupled with a decline in management. Significant losses in diversity and increases in competitive species

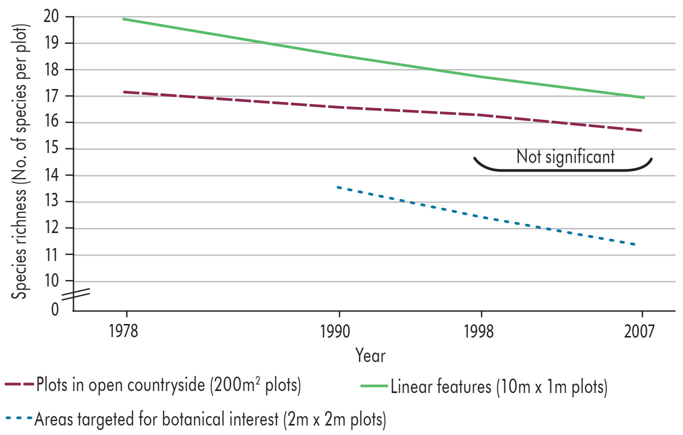

Figure A.4.1.4.1 Average species richness of vegetation in plots in the open countryside (fields, woods, heaths and moors), linear features, and areas targeted for their botanical interest in GB between 1978 and 2007. A decline in species richness is apparent in each dataset. Source: Carey *et al.* (2008). Countryside Survey data owned by NERC – Centre for Ecology & Hydrology.

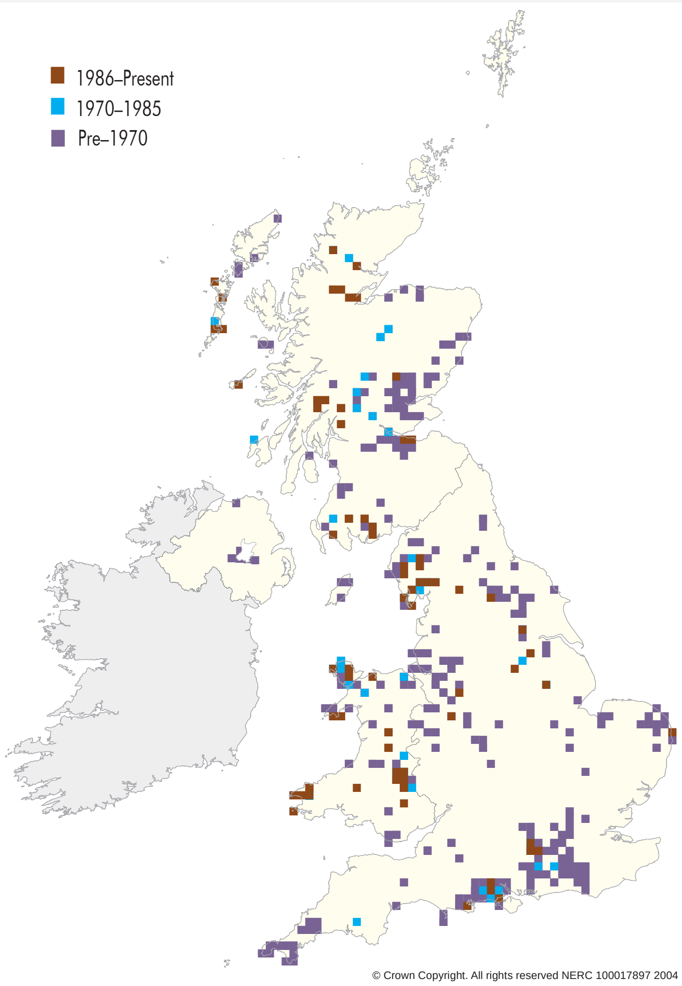

Figure A.4.1.4.2 Map showing distribution of *Pilularia globulifera* (pillwort) in mainland UK at 10x10 km resolution, based on 113 recent records (1986–present), 82 records from 1970 to 1985, and 544 historical records (pre-1970). Change index = -0.03. Most of the losses were pre-1970 and many date back considerably further. Source: data from the National Biodiversity Network Gateway (NBN 2011).

were found across GB between 1978 and 2007, especially along linear features (hedges, streamsides) and in small habitat patches (Carey *et al.* 2008b). Atmospheric nitrogen deposition has been shown to have a significant impact on plant species richness through direct toxicity, acidification and eutrophication (Stevens *et al.* 2004, 2010; Maskell *et al.* 2010; RoTAP 2011).

Global transport networks mix previously isolated biota, exposing more ecosystems to a greater number of potential colonists. There is concern about the impacts of non-native species on native plant diversity. The Atlas showed that some non-natives, such as giant hogweed (*Heracleum mantegazzianum*) (change index +2.0), Himalayan balsam (*Impatiens glandulifera*) (change index +1.85) and common rhododendron (*Rhododendron ponticum*) (change index +1.83), are increasing (Preston *et al.* 2002). Non-natives have been shown to have significant localised effects, but in the wider countryside across GB they are still relatively uncommon (Maskell *et al.* 2006). Invasion is likely to be facilitated by anthropogenic disturbance (intensive farming, atmospheric pollution and land use change), which increases the suitability of a habitat for a smaller number of 'winning' species. It has been shown that as species diversity declines there is an increase in functional similarity as winning trait syndromes dominate (Smart *et al.* 2006). This may have implications for the resilience of communities in providing ecosystem services.

Native vegetation is vulnerable to the increasing spread of non-native invasive fungal pathogens, such as Sudden Oak Death, *Phytophthora ramorum*, possibly linked with climate change. Upland vegetation, including woodland, has been seriously impacted by overgrazing by sheep and deer, resulting in a loss of biodiversity due to habitat fragmentation and the prevention of establishment.

Prospects: The public availability of distributional data (Preston *et al.* 2002) and its comparison with the information from the earlier Atlas (Perring & Walters 1962) has encouraged further data gathering that will be valuable in informing future trends. The BSBI is now recording in decade-long date classes (since 2000), and also aims to get tetrad maps of all species by 2020. Countryside Survey data is also publically available (www.countrysidesurvey.org.uk/) and now provides a 30-year time-series of data for common British plant species and habitats.

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A.4.1.5 Invertebrates

Marine and estuarine invertebrates

Author: Paul J. Somerfield

Taxa included in this group: Phyletic diversity is much higher in the sea than it is on land, and marine and estuarine invertebrates belong to most known animal phyla. The most abundant tend to be annelids (worms), molluscs (snails, clams), crustaceans and echinoderms (starfish, sea urchins).

Diversity in the UK: With a long and varied coastline, and a long history of investigation, UK coastal waters and estuaries are among the best studied in the world. Habitat diversity is high and some 9,000 species have been recorded.

Roles in ecosystem services: Many marine and estuarine invertebrates are harvested for food (crabs, lobsters, cockles, clams, oysters, scallops, cuttlefish, shrimp, prawns, urchins) and there is also a large industry harvesting them for bait (worms, crabs). Aquaculture is increasing; invertebrates form structures which provide habitats for commercial fish, and most commercial fish species feed on invertebrates. It should be noted that many marine invertebrates impose economic costs as well, especially those that form fouling communities on ships and structures. Marine sediments form the largest habitat in the UK and are a sink for human waste. The chemistry and functioning of the benthic system is often driven by the activities of invertebrates which turn over the sediment, move material from the surface to depth and vice versa, alter oxygenation, breakdown and recycle organic matter and nutrients, sequester pollutants or make them bioavailable, filter and cleanse seawater, and influence sediment mobility and stability. Habitat formation and modification are both performed by marine and estuarine invertebrates. Apart from their commercial importance, marine and estuarine invertebrates matter to people. Areas rich in invertebrate diversity are popular with scuba divers, and a day on the beach would not be complete without seashells.

Status and trends: For the determination of trends, data from some sort of standardised sampling through time is required. There are very few long- or even medium-term time-series for invertebrates in marine or estuarine environments; most of those available sample plankton. For other marine and estuarine invertebrates there are incomplete time-series available for sediments off the coast of Plymouth and Newcastle. Some studies of invertebrates on hard substrata have demonstrated changes over periods of time, including the fact that some intertidal communities are still recovering from the severe winter of 1962–1963, showing that recovery may be very slow. In general, however, our understanding of biodiversity change in estuarine and marine environments is poor, and much of what we know is based on scattered observations. The same is true of our understanding of status. Arguably, most marine systems were highly impacted before we began studying, and there is a whole branch of marine science attempting to reconstruct historical baselines. What is clear is that few or no marine systems are as they should be.

Drivers of change: The overwhelming factor influencing changes in the distribution and functioning of marine invertebrate biodiversity is human populations. Fishing has direct and indirect effects. It removes large predators which may induce changes in trophic structure throughout food webs. Removing fish which prey on urchins, for example, can lead to overgrazing and the development of urchin barrens instead of kelp forest. Demersal fishing has a direct effect on the benthos. Removal of filter-feeding invertebrates, such as oyster reefs, can alter water quality and benthic-pelagic coupling. Continual seabed disturbance alters the size, structure and composition of invertebrate communities, and hence their ability to cycle nutrients, sequester carbon and pollutants, and stabilise sediment. Removing biogenic (invertebrate) structure also removes habitat for juvenile fish.

Over long time periods, changes in land use and human activities impacts estuarine and marine invertebrates. Conversion of forests to agriculture, mining activity, canalisation of rivers and a range of other activities increase sediment loads in rivers, which either clog estuaries or alter sediment dynamics on coastal seas. Although direct disposal at sea of sewage sludge, untreated sewage, chemical waste and other things has reduced over recent decades, much of our waste still impacts estuaries and seas. Fertilisers applied to the land, petrochemicals and other contaminants, litter and plastics are all washed into rivers, where they join human waste heading for estuaries and the sea. Consequences include eutrophication, hypoxia, blooms of toxic algae, accumulation of persistent organic pollutants and disease. Habitat destruction heavily impacts coastal ecosystems. Marshes and mudflats are 'reclaimed', developments are built and channels are dredged (the dredgings of which are dumped). Many estuaries, particularly those with ports, are heavily modified. Replacement of soft (natural) sea defences, such as marshes, with hard structures, such as seawalls, removes habitat for invertebrates. Add to this sea-level rise (either natural, e.g. eustatic rebound, or anthropogenic, such as the consequences of melting ice) and coastal squeeze results. As a result, the intertidal habitat all but disappears (along with the services intertidal mudflats and marshes provide) and animals which feed on invertebrates, such as seabirds, waders and geese, have to feed elsewhere. Dredging alters hydrodynamics and sediment supply, and hard structures allow species which live on them to spread into areas where they did not live previously. Invasive species are transported on ships' hulls and in ballast water and can have deleterious consequences. Substances applied to prevent fouling by invertebrates can have unforeseen consequences. Tributyltin, for example, severely damaged populations of gastropods and bivalves. The consequences of climate change, such as increased storminess, sea-level rise, warming waters and acidification, may affect trophic interactions, larval production and survival, physiology and health.

Prospects: The overall trend in biodiversity in estuarine and marine invertebrates is unlikely to be an improving one in the foreseeable future. That being said, recent new legislation in the UK and the adoption of international and

European Union initiatives and directives suggest that the marine environment is being taken more seriously than it was. There is evidence that the careful management of the marine environment may lead to recovery of at least some of the biodiversity present in some marine environments. The move towards managing our seas in a more holistic, rather than sectoral, fashion is a start.

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Terrestrial and freshwater invertebrates

Authors: Allan Watt, Adam Vanbergen and Aidan Keith

Taxa included in this group: Terrestrial invertebrates include insects, spiders and other arthropods, snails, nematodes and earthworms.

Diversity in the UK: Terrestrial invertebrates comprise an estimated 95% of species globally. Knowledge of their diversity remains poor, even in the UK. There are, for example, approximately 50 species of woodlice in the UK, 50 centipedes, 60 millipedes, 650 spiders, 200 non-marine molluscs, 13 flatworms, 25 harvestmen, 260 springtails, 50 mayflies, 500 sawflies, 2,500 butterflies and moths, and 4,000 beetles. To put this in context, there are about 20,000 beetle species in Europe and 300,000 globally.

Roles in ecosystem services: Invertebrates play a major role in a range of ecosystem services, particularly as pollinators, natural enemies (predators and parasitoids) of agricultural and forest crops, and as decomposers. They may play a role in meaningful places and socially valued landscapes and waterscapes, but, apart from butterflies, this is probably minor in relation to other species groups.

Status and trends: There are long-term status and trend data for a range of invertebrate groups, including several insects groups (e.g. moths, butterflies, bees, dragonflies), spiders and some molluscs (e.g. land snails), but coverage is biased towards groups that are relatively easy to observe or trap, and groups that are culturally valued, particularly butterflies. As a result, data on these taxa provide limited, rather than comprehensive, information about trends in the invertebrate taxa that are responsible for the delivery of ecosystem services. Data on butterflies (**Figure 4.5**) and insect pollinators (Biesmeijer *et al.* 2006; Carvell *et al.* 2006) show significant recent population declines. The resampling of soil invertebrates as part of the Countryside Survey in 2007 (Emmett *et al.* 2010) established that the abundance of invertebrates in 2007 was greater than in 1998 under all broad habitats except arable (crops and weeds) (**Figure A.4.1.5.1**). The increased invertebrate abundance seen in 2007 was largely due to greater mite populations, suggesting that different soil groups may respond to environmental changes in a specific manner. The 2007 Countryside Survey also reported small, but statistically significant, reductions in the number of soil invertebrate broad taxa (**Figure A.4.1.5.1**) across a range of habitats.

Drivers of change: The key drivers of change in invertebrate abundance and diversity are land use and management, and pollution. There is also increasing evidence of the impact of climate change. Most of the evidence does not, however, relate to invertebrate groups that play a major role in the delivery of ecosystem services. Invertebrate natural enemies of pests have probably declined as a result of changes in crop species (Hicks *et al.* 2001) and agricultural intensification (Wilby & Thomas 2002; Tscharntke *et al.* 2005). Similarly, pollinators have suffered from changes to agro-ecosystems, including hedgerow removal (Hannon & Sisk 2009) and changes to field margins (Carvell *et al.* 2004).

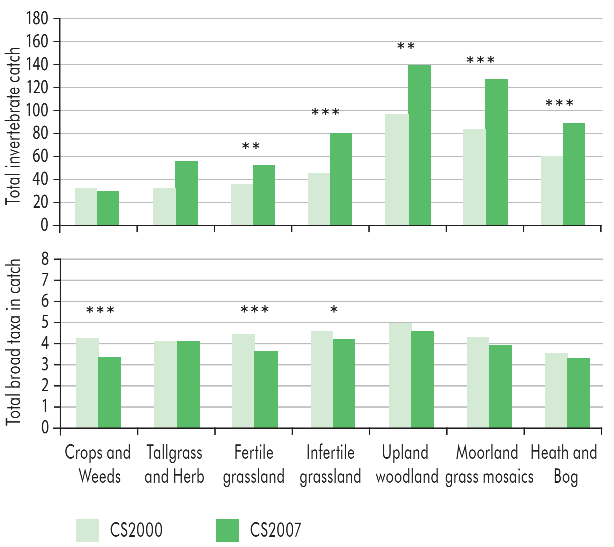

Figure A.4.1.5.1 Changes in soil invertebrate abundance and number of broad taxa recorded from 0–8 cm soil depth in 1998 and 2007 Countryside Surveys (CS2000 and CS2007); separated by Aggregate Vegetation Category. Asterisks indicate significant difference between surveys, * = P <0.05; ** = P <0.01; *** = P <0.001. Source: data from Emmett *et al.* (2010). Countryside Survey data owned by NERC – Centre for Ecology & Hydrology.

Prospects: Land use change is likely to continue to affect terrestrial invertebrates and climate change will probably have an increasing impact. Unless monitoring of invertebrates is expanded, however, our knowledge and understanding of trends in the abundance and diversity of terrestrial invertebrates will continue to be limited, particularly among the many taxa that play a role in the delivery of ecosystem services. Initiatives such as the Countryside Survey can provide useful information for some taxa; while such a single repeated sampling campaign cannot determine unequivocally whether change in soil invertebrate populations is underway, this kind of large-scale dataset is essential to understand potential drivers of change.

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A.4.1.6 Fish

Marine fish

Author: Simon Jennings

Taxa included in this group: Marine fishes, including transitional/diadromous species such as the European eel (*Anguilla anguilla*) and sturgeon.

Diversity in the UK: More than 330 fish species inhabit the shelf seas surrounding the British Isles. More than 13,000 species of marine fishes have been recorded globally.

Roles in ecosystem services: Marine fishes support fisheries and contribute to wild species diversity. Their presence can be enjoyed by recreational users of the sea such as divers and sea anglers. The first sale value of fish and shellfish taken by UK vessels from UK waters was £510 million in 2007 with an estimated Gross Value Added (GVA) of £204 million. Fish processing provided an additional GVA of £385 million in 2007. There are 12,729 full- and part-time fishermen in the UK. Participation in recreational sea angling was estimated to be 290,800 people from a boat and 480,950 people from the shore in 2007. The observation of marine fishes contributes to the diving experience and 190,000 recreational divers use the coast each year. The total expenditure by anglers resident in England and Wales in 2003 was estimated at £538 million, which consisted of £178 million for shore-based activities, £82 million for boat charters and £278 million for own-boat activities. Key sources: Frost (2010); Saunders (2010).

Status and trends: Since 1960, there have been major changes in fish populations and communities, primarily driven by the effects of fishing and climate. Trends in fishing mortality broadly reflected trends in fishing effort, with overall fishing mortality rates rising from 1960 to the mid-1990s, before stabilising and starting to fall in response to management measures. The marine environment has generally warmed since the 1960s, with the range of cold water species retracting northwards and warm water species invading from the south. The abundance of species in the cod family generally peaked in the late 1960s, owing to favourable environmental conditions, but subsequently fell, largely as a consequence of very high fishing mortality rates. Some species that were particularly vulnerable to fishing, owing to their low population growth rates, have decreased in abundance throughout the period. For instance, an especially vulnerable, but once abundant, species, the common skate (*Raja batis*), was regionally extinct by the 1970s. A recent taxonomic revision of this species implies that many of the remaining individuals that have been reported from outside the regions of extinction belong to a smaller and less vulnerable species. Thus the true common skate, named as the 'flapper skate' in this revision, may now be confined to very few areas and found only at very low abundance. Other vulnerable species include several deep-water fish species; sharks, rays and skates; and transitional/diadromous species such as the European eel and sturgeon. Many of these animals have recently been 'listed' as requiring statutory protection (OSPAR, CITES, Bern Conventions). As the abundance of many larger species has declined, so smaller species have proliferated, increasing the turnover time of communities. However, community-wide spatial patterns in fish diversity in UK waters, as measured with standard diversity indices that capture richness or evenness, reflect biogeographical factors as well as spatial variation in the effects of fishing. Key sources: Brander (1981); Pope & Macer (1996); Iglésias *et al.* (2009); Frost (2010); Saunders (2010).

Drivers of change: The main large-scale drivers of change in marine fish diversity are overexploitation and climate.

Prospects: In the last few years, there have been reductions in the number of overfished stocks and some signs of increases in the size of individuals in fish communities, but no signs of the recovery of the species most vulnerable to fishing. Thus, of 20 assessed finfish stocks in UK waters, the percentage thought to have full reproductive capacity and to be harvested sustainably has risen from 10% or less in the early 1990s to 20–30% in the 2000s and around 40% in 2007. The proportion of large individuals and species in fish communities has also started to rise in the last five years. The changes are thought to be a response to reductions in overall fishing mortality. Over the last eight years, European Union controls on fisheries have contributed to reductions in total fishing effort in the international demersal fisheries of around 30% or more in the North Sea, west of Scotland and the Irish Sea. During the last ten years, fishing mortality estimates for 67% of assessed fish stocks in UK waters have declined. The monitoring and assessment of fish stocks focuses on commercially exploited stocks and the bottom-dwelling fish community that lives over relatively smooth seabeds which can be sampled with survey trawls. As such, knowledge of changes in these fishes is better documented than changes in many estuarine, coastal, deep-water and highly migratory species, although some estuaries, such as the Thames and Severn, are well monitored using a combination of surveys and samples from power station cooling intake screens. Recent increases in the abundance of some commercial stocks and larger species have to be interpreted against a background of long-term depletion, and current levels of abundance and proportions of large species are still well below the highest levels recorded since 1960. The recent declines in fishing mortality rates appear to have little impact on the abundance of larger and more vulnerable non-target species. Key sources: Frost (2010); Saunders (2010).

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Freshwater fish

Author: Charles R. Tyler

Taxa included in this group: Freshwater fishes in the UK include members of the orders Cypriniforms, Acipenceriforms, Clupeiforms, Perciformes, Siluriformes, Anguilliformes, Atheriniformes, Mugiliformes, Salmoniformes, Esociformes, Gasterosteiformes, Petromyzontiformes and Gadiformes. Not all of these species reside for their full lives in freshwater, but rather move and/or migrate between freshwater and estuarine/marine environments, such as the European eel (*Anguilla anguilla*), Atlantic salmon (*Salmo salar*), river lamprey (*Lampetra fluviatilis*) and thick lipped grey mullet (*Chelon labrosus*).

Diversity in the UK: Sixty-four species of freshwater/brackish water fish have been recorded in the UK (www.wbrc.org.uk/worcRecd/Issue10/fishpopn.htm). Not all of these fish species are native, however, and some have been introduced either purposefully or accidentally. Purposeful introductions include the grass carp (*Ctenopharyngodon idella*) for weed control, rainbow trout (*Oncorhynchus mykiss*) for aquaculture and sport fishing, wels catfish (*Silurus glanis*) for sport fishing, and goldfish (*Carassius auratus auratus*) and orf, (*Leuciscus idus*) for aquaria and ornamental ponds. Accidental releases include the topmouth gudgeon (*Pseudorasbora parva*, sunbleak (*Leucaspius delineatus*) and pumpkin seed sunfish (*Lepomis gibbosus*). Worldwide, there are an estimated total of 26,000 species of fish in freshwater/estuarine and marine environments.

Roles in ecosystem services: Freshwater fishes are of considerable importance to UK ecosystem services and society as a whole. Wild UK freshwater fisheries have a significant economic value. As an example, commercial salmon and eel/elver fisheries in inland waters in England and Wales are thought to be worth up to £4 million annually (www.wbrc.org.uk/worcRecd/Issue10/fishpopn.htm). Recreational fishing, however, is the most important economic consideration for UK freshwater fisheries, and is estimated at £2.7 billion per annum, with an additional estimated £3 billion in the market value of fishing rights. The aquaculture of Atlantic salmon and rainbow trout are further significant economic enterprises in the UK, with annual tonnages in the region of 129,000 and 19,000, respectively (www.marlab.ac.uk/Uploads/ Documents/Survey00.pdf; Trout News 2005). Fish are also of considerable social and cultural importance in the UK. In historical times, commercial inland fisheries supported whole distinctive communities; today, there are approximately 3 million freshwater recreational anglers in England and Wales alone. In addition to their own conservation value, through recreational fisheries, fish ensure the endurance and protection of extensive freshwater habitats and their associated wildlife. Fish species and communities are arguably the best indicators of the well-being of aquatic ecosystems, in terms of both water quality and the physical environment.

Status and trends: In England and Wales there are regional differences in the trends for freshwater fisheries, but 13 species of fish are considered rare or threatened, and some, notably the burbot (*Lota lota*) and the common sturgeon (*Acipenser sturio*), are believed to be extinct in GB waters. Important commercial species in national decline include the Atlantic salmon, brown trout (*Salmo trutta fario*), grayling *(Thymallus thymallus*) and European eel. Considering the trends for Atlantic salmon in England and Wales, commercial catches have declined 40% in the last five years, but interpretation of these data is complicated by the fact that there have been increased regulatory controls and the buy-out of net licences during this time. Adult counts and returning stock estimates in UK rivers over available time-series show clear decline in some rivers (Itchen, Frome, Tamar and Thames); no substantive change in others (Dee, Test and Caldew); and an increasing trend in some (Tees, Fowey, Lune and Kent) (www.environment-agency.gov.uk/research/library/publications/33933.aspx). For eels, the picture is bleak: since the 1980s, the number of elvers (young eels) returning to European rivers has declined catastrophically to just 1% of their historic level; a decline clearly evident in the pattern of catches from the River Severn, the major elver fishery for England and Wales. Almost half of Scotland's 26 native fish species are thought to be declining including the river lamprey, brook lamprey (*Lampetra planeri*), allis shad (*Alosa alosa*), twaite shad (*Allosa falax*), Atlantic salmon, brown trout, Arctic char (*Salvelinus alpinus*), sparling (smelt; *Osmerus eperlanus*), European eel, and nine-spined stickleback (*Pungitius pungitius*). Eleven native species are considered threatened and one species, the vendace (*Coregonus albula*), has become extinct in Scotland. In contrast, pike (*Esox lucius*), minnow (*Phoxinus phoxinus*), roach (*Rutilus rutilus*) and perch (*Perca fluviatilis*) are increasing in Scotland (www.snh.org.uk/publications/on-line/advisorynotes/132/132.htm), favoured by warming due to climate change (Maitland 1991) and eutrophication (Maitland 1984).

Drivers of change: The main pressures affecting freshwater fish species throughout the UK include agricultural practices and habitat loss, pollution, and overfishing. The introduction of alien species, such as the zander (*Sander lucioperca*) which predates heavily on native species, and the topmouth gudgeon (*Pseudorasbora parva*) which can transmit diseases to native species, has impacted on specific freshwater fish populations. River engineering, habitat change and the creation of barriers to migration have also had an impact. The causes of the dramatic and widespread decline in eel populations in England and Wales (see below) are complex, but likely include changing ocean currents, loss of wetlands, disease, pollution and barriers such as dams and weirs (www.environment-agency.gov.uk/research/ library/publications /33933.aspx).

Prospects: Conservation management has brought about local recoveries for some fish species, such as the Atlantic salmon, particularly where water quality has been sufficiently improved (e.g. the Rivers Clyde and Carron). The implementation of restrictions on rod licences (and fish takes) and catch limits for netting have helped to protect the salmon and eel fisheries, but dissecting these factors from other contributing environmental factors is extremely complex. In England, pollution events that result in the decimation of wild local cyprinid fisheries have been (and still are) dealt with through restocking with fish supplied from a national breeding unit run by the UK Environment Agency. But the aim must be to reduce such pollution events through better regulation and control of environmental pollution discharges. Effective management and protection of UK freshwater fishes requires a better understanding of what drives their population dynamics—something that is lacking for almost all UK freshwater fishes.

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A.4.1.7 Amphibians

Author: Richard A. Griffiths

Taxa included in this group: Amphibian biodiversity in the UK comprises seven native species. Five of these species have widespread distributions: common frog (*Rana temporaria*), common toad (*Bufo bufo*), smooth newt (*Triturus vulgaris*), palmate newt (*Triturus helveticus*) and great crested newt (*Triturus cristatus*). The other two species are the natterjack toad (*Bufo calamita*), which has more specialist habitat requirements and is largely confined to sand dunes, lowland heath and saltmarsh habitats, and the pool frog (*Rana lessonae*), which is found at a single site as a result of a recent reintroduction.

Diversity in the UK: There are a number of areas within the UK where the five widespread species coexist and all five species may be found breeding within the same water body. Habitats supporting natterjack toad populations may also contain some of the widespread species, but it is uncommon to find six species occurring at the same site. All of the UK species are widespread in other parts of Europe, and are classified as 'Least Concern' on the IUCN Red List.

Roles in ecosystem services: The role of amphibians in ecosystem services is poorly understood, but frogs and newts frequently feature in urban conservation and green space initiatives. The establishment of garden ponds in urban and suburban areas has proved to be an effective way of establishing populations of some species and raising the profile of amphibians locally and nationally. Frogs and toads (and to a lesser extent newts) have featured prominently within literature and are of cultural importance within the UK.

Status and trends: All of our native species have suffered declines over the past 50 years. Since the 1960s, common frogs have made something of a comeback in urban areas through their ability to colonise small garden ponds quickly. Smooth newts—and to a lesser extent palmate newts and common toads—have made similar recoveries in some areas, although it is unlikely that such colonisations have offset declines within the wider countryside.

Drivers of change: For the widespread species, declines have been largely related to changes in agricultural practices that have resulted in the loss of breeding ponds and associated terrestrial habitat. Natterjack toads have suffered similar losses as a result of the development of coastal areas and heathlands for recreation, housing, agriculture and commerce (Beebee & Griffiths 2000). There is evidence that the isolation of populations as a result of development can lead to inbreeding depression. In addition, disease has become a recent issue for UK amphibians, with die-offs of some species regularly observed, particularly in garden ponds.

Prospects: Over the past two decades, interest in the conservation of amphibians has increased considerably within both the voluntary and professional sectors in the UK, and there are now several organisations that are carrying out conservation work, advocacy and public relations. Over the same timeframe, amphibian declines have become a high-profile conservation issue on a global scale, resulting in a range of initiatives to both highlight and tackle the problems. However, within the UK, amphibians frequently come into conflict with development and other activities involving changes in land use; national pressures for housing, food and commercial development are likely to result in the ongoing loss of populations despite mitigation and conservation interventions.

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A.4.1.8 Reptiles

Terrestrial reptiles

Author: Christopher Reading

Taxa included in this group: In the UK, the native reptiles comprise three snake species from two families (two from the Colubridae and one from the Viperidae) and three lizard species from two families (two from the Lacertidae and one from the Anguidae).

Diversity in the UK: The three snakes are the adder/viper (*Vipera berus*), the grass snake (*Natrix natrix*) and the smooth snake (*Coronella austriaca*). The three lizards are the common lizard (*Zootica vivipara*), the sand lizard (*Lacerta agilis*) and the slow-worm (*Anguis fragilis*). The smooth snake and sand lizard occur in southern England where they are at the northern edge of their geographical range. The grass snake occurs as far as northern England, but is absent from Scotland. The adder, slow-worm and common lizard occur throughout mainland GB. With the exception of the common lizard, there are no reptiles in Ireland.

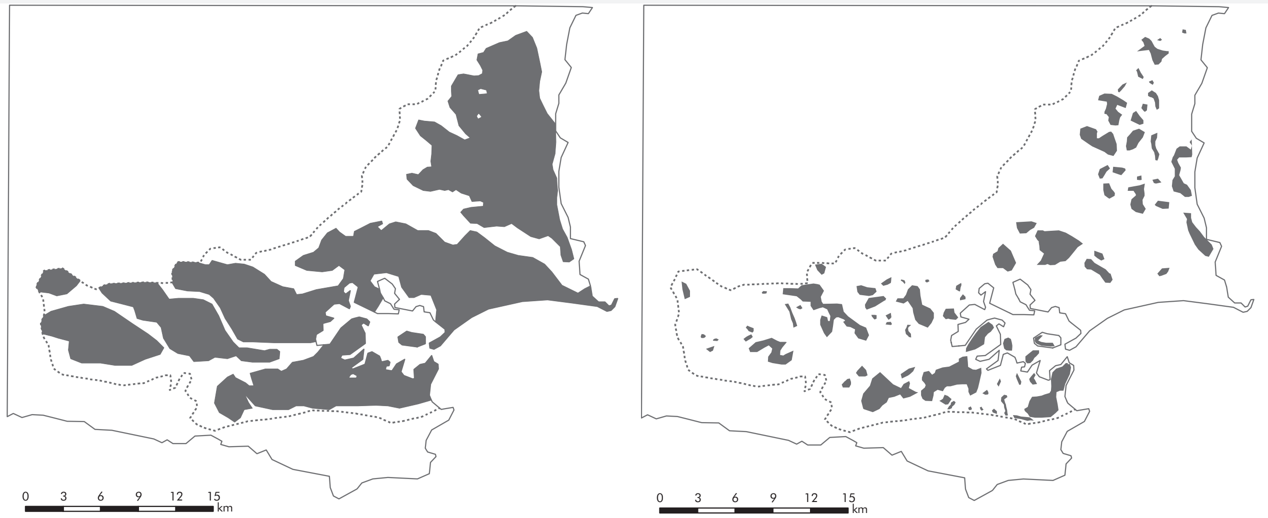

Figure A.4.1.8.1 Changes in the extent of lowland heathland in Dorset between a) 1759 and b) 1978. Source: Webb & Haskins (1980). Copyright (1980), reproduced with permission from Elsevier.

Roles in ecosystem services: With respect to trees and vegetation, the effect of the presence of reptiles relates to the preservation of the habitat where they occur. This is particularly true for the nationally rare sand lizard and smooth snake as the habitats in which they occur gain a significant measure of protection under the Wildlife and Countryside Act, 1981, due to their presence. This also applies to habitats where the other four species occur, though to a lesser degree. In this respect, the presence of rare reptile species can have a financial impact on the potential development of land as Environmental Impact Assessments are required to be completed, which may call for subsequent mitigation measures to be undertaken.

Status and trends: The two rarest species (sand lizard and smooth snake) are almost totally restricted to dry lowland heath in the south of England; being at the northern edge of their geographical range, change in the extent (**Figure A.4.1.8.1**) and area (**Figure A.4.1.8.2**) of this habitat type may be used as an indication of how the status of these two species has changed (declined) over time (Webb & Haskins 1980; Rose *et al.* 2000). With the exception of the smooth snake, there are almost no reliable long-term datasets for any of these species, preventing changes of status over time to be determined. The data that is available cannot be totally relied upon as it has been collected in many different ways, by people with varying degrees of expertise, making comparisons between recorders and between years of dubious value. In general, however, the consensus is that all six species are probably in decline, mainly as a result of habitat loss, though the slow-worm remains relatively widespread and locally abundant.

Drivers of change: The warming effects of climate change in the UK are likely to extend the potential range of all the species northwards. Nevertheless, reptile populations are likely to continue to decline due to habitat loss and disturbance—the adder appears to be particularly susceptible to disturbance.

Prospects: In the future, the main threat to all six species, but to the two rarest reptiles in particular, is likely to be habitat loss as the areas in which they occur are fragmented and lost to development. If the climate warms, as predicted, it is possible that some species may potentially extend their range northwards. However, any benefits from a warming climate are likely to be outweighed by habitat loss and disturbance.

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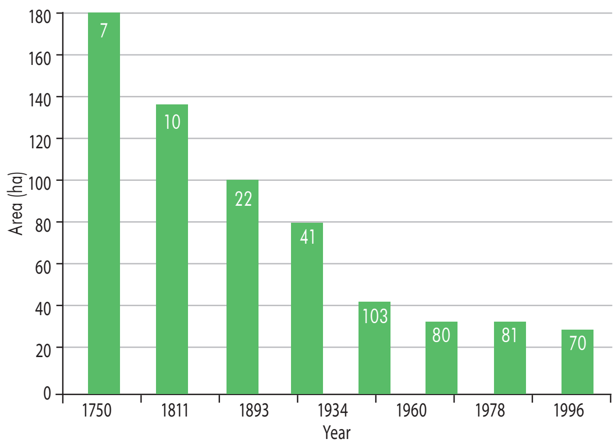

Figure A.4.1.8.2 The change in the area of heathland recorded in surveys between 1750 and 1996. The figures above the bars show the number of heathland fragments greater than 4 hectares that existed (data from the three ITE Dorset Heathland Surveys, 1978, 1987 and 1996, and historical data from Moore, 1962). Source: Rose *et al.* (2000). Copyright (2000), reproduced with permission from Elsevier.

Marine reptiles

Author: Matt Frost

Taxa included in this group and diversity in the UK: The only marine reptiles found in UK seas are turtles. Of the seven species of marine turtle found worldwide, four are known to occur occasionally in seas around the UK: leatherback turtle (*Dermochelys coriacea*); loggerhead turtle (*Caretta caretta*); Kemp's ridley (*Lepidochelys kempii*); and green turtle (*Chelonia mydas*). These species occur in extremely low numbers and only the leatherback turtle is seen frequently enough (an average of 33 records a year since 1998) to be considered a true member of the UK fauna (Marubini 2010). There are also two individual records (1953 and 1983) for hawksbill turtle (*Eretmochelys imbricate*), but this species is not considered a visitor to UK waters (Howson & Picton 1997).

Role in ecosystem services: Marine turtles play a role in marine ecosystems worldwide by "maintaining healthy seagrass beds and coral reefs, providing key habitat for other marine life, helping to balance marine food webs and facilitating nutrient cycling from water to land" (Wilson *et al.* 2010). Wilson *et al.* (2010) also found that, as regards direct human use, the non-consumptive use of turtles (mainly tourism) was of much higher value than consumptive use. In the UK, the value of marine turtles is mainly as flagship species used to promote interest in, and engagement with, the marine environment.

Status and trends: Six of the seven species of marine turtle are classified by IUCN as Endangered or Critically Endangered. The leatherback is listed as Critically Endangered and is also on the OSPAR list of threatened or declining species, along with the loggerhead turtle. On a global scale, there have been very large declines in numbers of marine turtles (Bjorndal & Jackson 2003), but the numbers of turtles in UK waters are too few to make any confident assessment of state or trend for this area (Marubini 2010).

Drivers of change: It is important to note that turtles are wide-ranging species; the leatherback turtle, in particular, migrates throughout the Atlantic, with UK waters representing only a small, peripheral part of its summer foraging habitat (McMahon & Hays 2006; Witt *et al.* 2007). Marine turtle populations are, therefore, affected by drivers operating at larger scales than just the UK level, with pressure from commercial fisheries, habitat loss and climate change being considered the main drivers of change globally (Bjorndal & Jackson 2003). In the UK, entanglement in fishing ropes and nets, and ingestion of plastic debris, constitute the main threats, but occurrences are rare and their impact at the population level has not yet been assessed (Marubini 2010). There is little evidence, as yet, of whether climate change will have any effect on the distribution of turtles in UK waters, although leatherback turtles are expected to expand their range into higher latitudes as ocean temperatures increase (McMahon & Hays 2006).

Prospects: Marubini (2010) recommends that an international effort be made around the entire western approaches to the European shelf (with a focus around the Bay of Biscay) to estimate numbers and trends in marine turtles and inform conservation efforts undertaken in the UK and globally. This is extremely important as species of marine turtle, including the leatherback, remain Critically Endangered and at the risk of ecological and, in some cases physical, extinction (Wilson *et al.* 2010).

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A.4.1.9 Birds

Authors: Richard D. Gregory, Richard B. Bradbury

Taxa included in this group: Here, we consider all birds (Vertebrata: Aves) occurring naturally in the UK.

Diversity in the UK: Around 250 bird species occur naturally in the UK on a regular basis, as resident or summer breeders, or as wintering or passage migrants (Gibbons *et al.* 1996; Gregory *et al.* 2002; Eaton *et al.* 2009). About 85% of these species breed in the UK, of which, 20% are rare breeding species (fewer than 300 pairs). In terms of global significance, the UK is home to large fractions of the global populations of a range of breeding seabirds and wintering wildfowl and waders.

Roles in ecosystem services: It can be argued that birds play a major role in wild species diversity, meaningful places, and socially valued landscapes and waterscapes. Large, enigmatic, flagship bird species hold a special fascination and attraction for people, as do garden birds (Crocker & Mabey 2005). The latter often introduce people to nature for the first time and represent their most common interaction with wildlife. Interest in birds is reflected, for instance, in over half a million people participating in garden birdwatches each January; over a million people being members of the Royal Society for the Protection of Birds (RSPB); and very substantial spending on bird-feeding.

Status and trends: Bird numbers and geographical ranges are tracked by a variety of survey schemes (Gibbons *et al.* 1996; Gregory *et al.* 2002; Eaton *et al.* 2009). Population trends of all but the rarest species are captured in multi-species 'Quality of Life' indicators (**Table A.4.1.9.1; Figure A.4.1.9.1**). The trend for all species with adequate data is relatively stable over four decades, but average trends differ according to the main habitat of the species (**Table A.4.1.9.1**). On average, seabird populations have increased, but they are now in decline. Birds associated with wet breeding habitats show population stability. Woodland and farmland birds have declined markedly, and while the former show greater stability in the last decade, the latter do not. Urban birds have increased over the last decade or so (**Table A.4.1.9.1**). Similar information is not available for the UK uplands, but some wading birds and songbirds at least appear to be in decline (Sim *et al.* 2005). Within habitats, generalist birds have tended to prosper, while specialists have declined. Among a smaller number of rare breeding species (40), occupying various different habitats, around 60% of have increased in number in recent decades. These include charismatic birds such as red kite (*Milvus milvus*), marsh harrier (*Circus aeruginosus*), stone-curlew (*Burhinus oedicnemus*), woodlark (*Lullula arborea*) and Dartford warbler (*Sylvia undata*). Wintering waterbirds have increased substantially in recent decades, but have declined in the most recent decade for which there is data (**Table A.4.1.9.1**).The number of bird species of high conservation concern in the UK has risen steadily over the last two decades (**Table A.4.1.9.2**). The latest assessment identifies 52 species in the highest category of conservation concern, mostly because of population declines.

Drivers of change: Evidence to link the decline of farmland birds with changes in agricultural practices is compelling (Wilson *et al.* 2009). There is also evidence linking change in woodland structure—itself driven by changes in forestry management, forest maturation and increased deer-browsing—with woodland bird populations (Hewson *et al.* 2007). Trends among breeding waterbirds are less well understood and seem to be linked to agricultural intensification and, perhaps, to the predation of ground-nesting birds (Ausden *et al.* 2009). Several of these species are long-distance migrants, with wintering grounds south of the Sahel, so numbers here may be driven by factors on their wintering grounds or migration sites. However, evidence for such effects is limited. Seabird numbers are linked in a complex fashion with fishery practices, marine food chains and oceanic changes (JNCC 2009). Increased discards may, in part, be responsible for the rise of seabird numbers in recent decades. Growing numbers of urban birds may be linked to wildlife-friendly management of green space and gardens, and increased food provision in gardens. Climatic change is frequently cited as a potential driver of trends in birds and there is increasing evidence for impacts in different habitats and speculation about its potential effects. Climatic change might be benefiting southern bird species, but acting to the detriment of those whose southern boundary lies in the UK (Green *et al.* 2008; Gregory *et al.* 2009). Climate change will interact with, and exacerbate, other drivers.

Prospects: The general loss of bird populations, due to the drivers discussed above, will diminish the delivery of wild species diversity, meaningful places, and socially valued landscapes and waterscapes, all of which are enriched by birds. However, increasing numbers of gardens birds, and the recovery and increase in endangered and charismatic bird species, most often through intensive conservation programmes, will increase delivery of the same range of services.

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Table A.4.1.9.1 UK 'Quality of Life' indicators: population trends of wild bird species in different habitats. UK Sustainable Development and England Biodiversity Strategy Indicators. Source: data from the RSPB, BTO and Defra (2010).

Species group (number of species)	Long-term trend	Short-term trend	Key drivers
Breeding birds	**1970–2008**	**1998–2008**	
All species (114)	3%	6%	Multiple and diverse
Seabird species (19)	28%	-5%	Fishery practice and oceanic change
Water and wetland species (26)	1%	9%	Change in agricultural practices
Woodland species (38)	-14%	5%	Change in woodland structure
Farmland species (19)	-47%	-4%	Change in agricultural practices
Urban species (27)	-	11% *	Sympathetic management and food provision
Wintering birds	**1975/1976–2006/2007**	**1996/1976–2006/2007**	
All waterbird species (46)	57%	-6%	Site and species protection and management
Wildfowl species (27)	62%	-9%	
Wader species (15)	44%	-5%	

* English trends 1994–2008.

Table A.4.1.9.2 Number of bird species of high, medium and low conservation concern.

Year	High concern	Medium concern	Low concern	
1990	117		Not assessed	Batten *et al.* (1990)
1996	36	110	Not assessed	Gibbons *et al.* (1996)
2002	40 (16%)	121 (49%)	86 (35%)	Gregory *et al.* (2002)
2009	52 (21%)	126 (51%)	68 (28%)	Eaton *et al.* (2009)

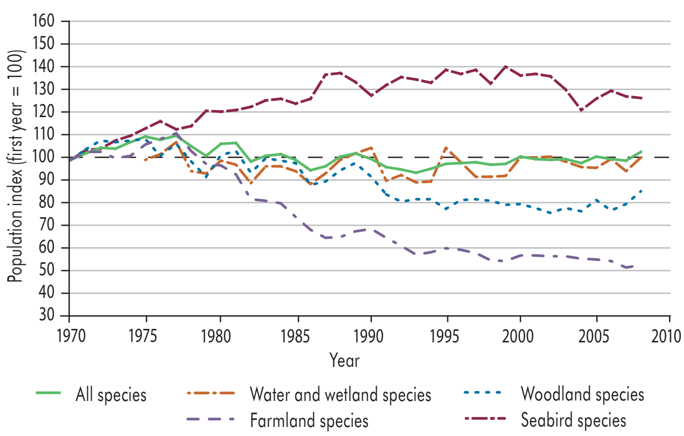

Figure A.4.1.9.1 UK 'Quality of Life' indicators: Population trends of wild birds. The graph shows the composite population trends of UK breeding bird species (n=114) with subdivisions showing grouped species' trends for seabirds (n=19), water and wetland birds (n=26), woodland birds (n=38), and farmland birds (n=19). On average, populations of woodland and farmland birds have fallen between 1970 and 2008 by 14% and 47% respectively. Source: data from RSPB, British Trust for Ornithology, JNCC and Defra.

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A.4.1.10 Mammals

Terrestrial mammals

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Taxa included in this group: This section includes both native and non-native UK species, as well as domestic livestock, which, although not present in the 'wild', may have significant impacts on the wider environment.

Diversity in the UK: There are 62 species (40 native and 22 non-native) and 4 subspecies of wild terrestrial mammal in the UK (Tracking Mammals Partnership, www.jncc.gov.uk/page-1757). Four species are on the IUCN Red List (IUCN 2009: the Scottish wildcat (*Felis silvestris*), which is listed as Vulnerable; and the otter (*Lutra lutra*), Bechstein's bat (*Myotis bechsteinii*) and barbastelle bat (*Barbastella barbastellus*), all of which are categorised as Near Threatened. The main mammalian livestock species present are beef and dairy cattle, sheep and pigs (Defra 2009).

Roles in ecosystem services: Terrestrial UK mammals have very important impacts upon: crops, livestock and fish through herbivore browsing and grazing (Wallis De Vries 1995; Putman & Moore 1998), potential disease transmission (e.g. badgers and TB; Macdonald *et al.* 2006) and predation (e.g. the potential impact of otters on local fisheries; Kruuk 2006); wild species diversity, primarily through their presence as a conspicuous component of our wild diversity, but secondarily through grazing (essential for the maintenance of certain habitats of high biodiversity value (Defra 2009)) and predation (by invasive mammal species in particular, for example, American mink *Mustela vison*, predate water voles *Arvicola terrestris*, and seabirds; Macdonald & Harrington 2003); meaningful places and socially valued land and waterscapes through their intrinsic existence value (White *et al.*1997; Dutton *et al.* 2010). Mammals have medium importance impacts upon: trees, standing vegetation and peat, principally through grazing actions (Thalen 1984), but also potentially through the ecosystem engineering effects of beavers (*Castor fiber*) (Rosell *et al.* 2005); and climate regulation, primarily through the production of the greenhouse gas methane by livestock (particularly cattle) (Defra 2009). Finally, terrestrial mammals may impact water purification through the ecosystem engineering action of beavers (e.g. Balodis 2004).

Drivers of change: The drivers of species trends for wild mammals in the UK include habitat- and climate-related factors (such as population increases and increasing urbanisation), human management ('pest' species control, reintroduction, the introduction of invasive species, and protection of species of conservation concern) and natural opportunities (i.e. habitat availability and a lack of predators, for example, for sika deer, *Cervus nippon*). But in many cases trends are unknown (e.g. the hedgehog *Erinaceus europaeus*) or not completely understood (e.g. the American mink). Because there are no common drivers for trends in wild mammals in the UK, and because the trends themselves vary among species, it is difficult to identify 'indicator' species for this group. The table below, therefore, describes recent trends, and their drivers, for a number of 'example' mammals. Drivers of trends in domestic livestock numbers are largely economic (including market prices and agri-environment schemes), but also include new regulations (see, for example, Defra 2009).

Status and trends: Unlike some taxa (e.g. bird species, the majority of which are relatively visible when present), wild mammal species are not always amenable to direct survey. Many species are cryptic and/or nocturnal, and their presence can only be inferred from indirect signs such as scats, footprints, homes (burrows, dens, nests, etc.), spoils (such as molehills or leftovers), or by invasive means such as live trapping. For this reason, nationwide surveys of wild mammal species in the UK are a rare and relatively recent phenomenon; they tend to be species-specific and omit a large number of species. The following information on wild mammal species trends is summarised from the Tracking Mammals Partnership (TMP) report (Battersby 2005), which covers 37 (57%) UK terrestrial mammals. The TMP report highlighted the fact that there are still insufficient data for approximately half of terrestrial mammals. Sufficient data were available to make some assessment of population change for 33 species and one subspecies.

Of the 24 native wild species and one native subspecies, 40% appear to be increasing, 12% declining and 16% remain stable. There were insufficient data to assess population trends for the remaining 32%. Native species currently increasing include several of the bat species, red and roe deer (*Cervus elaphus* and *Capreolus capreolus*) and several carnivore species (polecat *Mustela putorius*, badger *Meles meles*, and otter). Declining native species include the water vole, the dormouse (*Muscardinus avellanarius*) and the hedgehog (**Table A.4.1.10.1**).

Of the nine non-native wild species, 66% appear to be increasing, 11% declining and 22% are stable. Non-native species that are currently increasing include the grey squirrel (*Sciurus carolinensis*), the brown rat (*Rattus norvegicus*), and sika, fallow (*Dama dama*), Chinese water (*Hydropotes inermis*) and muntjac (*Muntiacus reevesi*) deer. The brown hare (*Lepus europaeus*) appears to be stable and the American mink appears to be declining. In most cases, trends appear to be unchanged over the last 25 years, except for the rabbit (*Oryctolagus cuniculus*) and the American mink, both of which appear to have declined in recent years following earlier increases, and for red foxes (*Vulpes vulpes*), red and fallow deer, for which earlier increases appear to have stabilised. For bats, there are insufficient data available to assess longer-term trends.

Over the last 25 years, the UK holdings of key mammalian livestock groups, e.g. cattle, sheep and pigs, have declined (Defra 2009).

Prospects: Prospects for UK mammals may differ greatly between wild and domestic species. Future numbers of domestic mammal species will respond to global and national economics, human population trends and changes in agricultural policy, but may also be affected by climate change, and so, are difficult to predict. Future prospects for wild mammals are most likely to be species-specific rather than general across all taxa. For established non-native species, changes in their populations or geographical range may occur as a result of human intervention (e.g. a national programme for the eradication of American mink or grey squirrels, although none are currently planned). The national population and range of several native species of conservation concern may increase due to reintroduction programmes (e.g. water voles), control programmes or due to strengthening of legislation (in particular the Wildlife and Countryside Act). Additionally, reintroductions may include previously extirpated species, such as beaver and lynx (*Lynx lynx*), so have the potential to increase mammalian biodiversity in the UK.

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Table A.4.1.10.1 Examples of trends in mammals (25 years to 2007).

	Common name	Scientific name	Trend	Drivers
Native wild species (by Order) (JNCC 2007)				
Insectivora	Hedgehog	*Erinaceus europaeus*	↓	Unknown
Chiroptera	Common pipistrelle	*Pipistrellus pipistrellus*	↑ (last 10 years)	Increased survival during hibernation (through warmer winters) and increased recording effort?
Lagomorpha	Mountain hare	*Lepus timidus*	No significant evidence	Not applicable
Rodentia	Water vole	*Arvicola terrestris*	↓	Habitat fragmentation and predation by American mink
Carnivora	Otter	*Lutra lutra*	↑	Improvement in water quality since organophosphates banned, end of persecution and reintroductions
Artiodactyla	Roe deer	*Capreolus capreolus*	↑	Increased availability of habitat (through afforestation) and forage (through planting of winter crops)
Non-native wild species (by Order) (JNCC 2007)				
Lagomorpha	Rabbit	*Oryctolagus cuniculus*	↑ (long-term) / ↓ (recent)	Unknown
Rodentia	Brown rat	*Rattus norvegicus*	↑	Increased urbanisation / changes in refuse quantities and collection?
Carnivora	American mink	*Mustela vison*	↓	In part mink removal, potentially other, unknown, factors (otter recovery has been suggested as a driver but the evidence is equivocal)
Artiodactyla	Reeve's muntjac	*Muntiacus reevesi*	↑	Natural dispersal into unoccupied areas and lack of predation in the UK
Domesticated livestock (by Family) (Defra 2009)				
Bovidae	Cattle	*Bos primigenius*	↑ (last 10 years)	Economic and policy drivers
Suidae	Pigs	*Sus domestica*	↓ (last 10 years)	
Bovidae	Sheep	*Ovis aries*	↓ (last 10 years)	

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Marine mammals

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Taxa included in this group and diversity in the UK: The marine mammal groups found in UK waters are the whales, dolphins and porpoises (collectively known as cetaceans) and the seals (pinnipeds). The otter (*Lutra lutra*) is also found in sea lochs and coastal environments but, as a semi-aquatic mammal, is not considered further here. There are 28 species of cetacean recorded in UK waters, which is a high level of diversity considering the UK's comparatively small proportion of the North Atlantic (Pinn 2010). Of these, 11 are known to occur regularly, while the remaining 17 species are considered to be vagrants or rare visitors (Pinn 2010). Although only two species of seal live and breed in the UK, they are of international importance with approximately 36% of the world's population of grey seals (*Halichoerus grypus*) and 4% of the world's population of harbour seals (*Phoca vitulina*; also known as common seals) found in the UK (Duck 2010). The largest populations are in Scotland which has 90% of the UK's population of grey seals and 80% of the harbour seals (Duck 2010). Five species of Arctic seal infrequently visit UK waters (Hall 2008).

Role in ecosystem services: Marine mammals play a key role in the marine ecosystem as top predators and are able to have a major influence on the structure and function of some aquatic communities (Bowen 1997). The main human value for marine mammals in the UK, as in many parts of the world, is in ecotourism and as 'flagship species', defined as: "popular charismatic species that serve as symbols and rallying points to stimulate conservation awareness and action" (Leader-Williams & Dublin 2000). Beaumont *et al.* (2008) point out that marine mammal biodiversity is very important for UK tourism with whales and dolphins being Scotland's number one wildlife attraction and the value of seal-watching to the UK economy in 1996 being at least £36 million.

Status and trends: The conservation status of cetaceans in the eastern North Atlantic has recently been assessed under the requirements of the Habitats Directive. The status of five species is considered favourable. The status of a further six species is unknown, due to a lack of data, while the remaining 17 species are either rare or vagrant so their conservation status in UK waters could not be assessed (Pinn 2010). The grey seal population has steadily increased since routine monitoring started in the 1960s. The increase in pup production is at least partly due to the availability of new breeding sites following the abandonment of human settlements on remote islands, including the automation of lighthouses (Duck 2010). Grey seal pup production is now stabilising, probably due to density-dependent factors affecting the general population. In contrast, harbour seal numbers have declined significantly in a number of areas, with populations in Shetland, Orkney and the on east coast of Scotland declining by more than 50% since 2001; the causes of these declines are not yet known (Duck 2010).

Drivers of change: As wide-ranging migratory species, drivers affecting many cetacean populations operate at a scale larger than just the UK. Marine mammals are apex predators and anthropogenic pressures (mainly commercial hunting and persecution) have had the largest impact on populations in the past (commercial whaling was only banned by the International Whaling Commission in 1986). For UK cetaceans, direct mortality through bycatch in fishing gear remains the most important human impact, with dolphins and porpoises being particularly vulnerable. For seals, since culling ended in the 1970s, the main impacts on UK populations have affected harbour seals. Two outbreaks of Phocine Distemper Virus (PDV) in 1988 and 2002 reduced harbour seal populations, particularly on the east coast of England, by 50% and 22% respectively. Between 2001 and 2009, harbour seal populations in Shetland, Orkney and on the east coast of Scotland declined by up to 60%, the causes of which are unknown.

Prospects: Predicting future trends for UK marine mammals is very difficult for a number of reasons. It is not possible for cetaceans due to uncertainties in the relationship and influence of pressures on population dynamics (Pinn 2010). For seals, although it is thought that the grey seal population is likely to stabilise over the next decade, it is difficult to predict future trends in harbour seals because the causes for recently observed declines have not yet been determined and the future impact of PDV is unknown.

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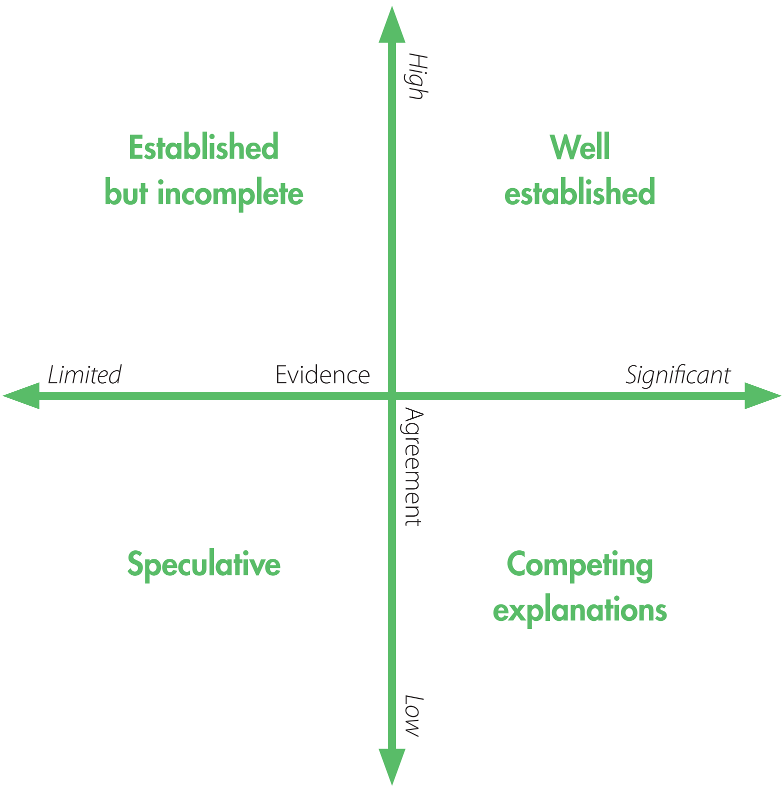
Appendix 4.2 Approach Used to Assign Certainty Terms to Chapter Key Findings

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.