Chapter 13:
Supporting Services

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Key Findings*

Supporting services underpin the delivery of all other ecosystem services. Therefore, understanding their response to key drivers, such as climate change, land use and nutrient enrichment, is vital for the sustainable management of the UK's land and water resource. Supporting services include the ecological status of soil and water, and processes that drive the formation of soils, cycling of nutrients and fixation of carbon by plants. These are all strongly interrelated and, in many cases, underpinned by a vast array of physical, chemical and biological interactions. Our understanding of the ways that these interactions influence supporting services, and of the relative contribution of biological, chemical and physical factors, is generally limited.

The soils of the UK are diverse and relatively young as most of Britain was under ice or peri-glacial conditions until 10–15,000 years ago. Soils form slowly, but can be quickly degraded and lost[1]. Soil formation rates in the UK are 0.04–0.08 mm per year for mineral soils, which is less than 1 cm per century, although there is still a lack of data with which to formulate a truly accurate picture[2]. In actively growing bogs of good habitat status, peat formation is approximately 0.8 mm/yr, which is equivalent to a carbon accumulation rate of 0.5 tonnes carbon/hectare/year (t C/ha/yr). However, the average is probably closer to 0.1 t C/ha/yr given that many peatlands are not of good status[2]. Threats to soil formation include organic matter loss due to climate warming, inundation of coastal soils due to sea-level rise, erosion and compaction caused by intensive agriculture, and soil-sealing due to urbanisation[1]. There are many consequences of loss of soil for supporting, regulating and provisioning services.

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

The last 50 years have witnessed substantial changes in the nutrient status and pH of waters and soils, with likely consequences for the delivery of both regulating and provisioning services. One of the most dramatic trends in nutrient cycling has been the enrichment of terrestrial and aquatic habitats with nitrogen due to the use of nitrogen fertilisers to increase food production[1]. This has resulted in substantial changes in plant productivity, plant species diversity and composition, and an accelerated rate of nitrogen cycling[1]. Another major change in soils and waters is the recent decrease in acidity of surface soils and acid-sensitive waters due to a substantial decline in sulphate deposition and an increase in rainfall pH since the late 1970s[1].

[1] *well established*

Recent evidence suggests that there may have been a widespread decline in the availability of phosphorus across terrestrial and aquatic systems over the last decade. Across UK terrestrial systems, extractable phosphorus in soils has apparently declined by an average of 25% between 1998 and 2007[1]. The reasons for this are not understood and require further investigation. There is evidence of decreasing phosphate concentrations in rivers across the UK which is linked to a reduction in the application of phosphorus fertiliser to land[1].

[1] *well established*

* Each Key Finding has been assigned a level of scientific certainty, based on a 4-box model and complemented, where possible, with a likelihood scale. Superscript numbers and letters indicate the uncertainty term assigned to each finding. Full details of each term and how they were assigned are presented in Appendix 13.1.

Spatial variation in rainfall and runoff is exceptionally high across the UK and few general trends in precipitation and annual runoff have been identified. However, a change in rainfall seasonality has been observed, with wet winters being more common in the last 30 years[1]. A gradual increase in annual average evaporation loss from 500 mm to 550 mm during the period 1980 to 2005 has been observed[1]. This probably reflects an increase in average temperatures across the UK of around 1.0°C over the last 30–40 years.

[1] *well established*

There is significant inter- and between-year variation in terrestrial primary production that is driven primarily by weather patterns, such as those which occurred during the summer drought of 2003[1]. There is mounting evidence that climate change will impact on primary production and community composition across all UK habitats, but there is an even greater risk of dramatic changes occurring as a result of extreme weather events[2]. Changes in primary production resulting from climate change will have significant implications for provisioning and regulating services.

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

In agricultural and forestry systems, improvements in land management have had a major impact on primary production due to nutrient input, technological developments and genetic selection[1]. This is reflected in a ten-fold increase in yields in some agricultural systems over the last century. In semi-natural systems, atmospheric nitrogen deposition has increased primary productivity[2]. In lakes, streams and coastal waters, nutrient inputs of nitrogen and phosphorus from sewage and fertiliser runoff have caused major increases in primary production, in some cases eliminating seasonal patterns and causing undesirable hypoxia[1].

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

In general, there is much uncertainty about the mechanisms that underpin supporting services, and our knowledge of how these services will be affected by current and future drivers, including climate change, is limited. However, it is clear that the mechanisms that underpin supporting services vary greatly across habitats, and the effects of key drivers, such as climate change, land use and nitrogen deposition, will impact on supporting services differently across UK habitats. Research is urgently needed to develop sustainable options for the management of UK supporting services and the regulating and provisioning services that they underpin.

13.1 Supporting Services and Human Well-being

Supporting services provide the basic infrastructure of life, including the capture of energy from the sun, the formation and maintenance of soils for plant growth, and the cycling of water and nutrients in terrestrial and aquatic ecosystems. In other words, supporting services are those that are required for the production of all other ecosystem services (i.e. regulating (Chapter 14), provisioning (Chapter 15) and cultural services (Chapter 16)), and include ecosystem functions of primary production, decomposition and nutrient cycling, water cycling and the formation of soils (EASAC 2009). Supporting services differ from other ecosystem services in that their impacts on human well-being are indirect and mostly long-term in nature; in the context of the UK NEA conceptual framework, they are the primary and intermediate services that underpin the final ecosystem services that are more directly linked to goods (Chapter 2). For example, soil formation involves changes in the physical, chemical and biological properties of soil over decades, centuries and even millennia (Jenny 1941), and its impacts on human well-being are mostly indirect, influencing regulating and provisioning services. Likewise, nutrient cycling is driven by many abiotic and biological factors that operate over both short and long timescales (Chapin *et al.* 2002; Vaughan *et al* 2009), and its impact on human well-being is indirect, affecting regulating and provisioning services.

Supporting services are all strongly interrelated and, in many cases, they are underpinned by a vast array of physical, chemical and biological interactions (**Figure 13.1**; Acreman *et al.* 2009; Bardgett & Wardle 2010). For instance, primary production both influences, and is influenced by, the supporting services of nutrient cycling, water cycling and soil formation. Moreover, although primary production is strongly regulated by external nutrient inputs and various environmental pressures (e.g. land use, climate change and atmospheric pollution), it is also dependent, in part, on the biological composition of communities and biotic interactions, for example between plants, plants and soil organisms, and between plants, herbivores and their consumers (Bardgett and Wardle 2010). Indeed, the last two decades have witnessed an explosion of interest in the roles that species and their interactions play in ecosystems, to the extent that this topic is now a dominant theme in ecology. This has been motivated by many factors, including a growing recognition of the roles that organisms and their interactions play in driving ecosystem processes (Grime 1979; Lawton and Jones 1995; Wardle *et al.* 2004; Bardgett & Wardle 2010), and a vast research effort aimed at connecting biodiversity, in terms of species richness and composition, to ecosystem functioning (Hooper *et al.* 2005; Naeem *et al.* 2009). Furthermore, the growing interest in predicting how species' responses to global change influence supporting services, such as decomposition, nutrient cycling and primary production, has generated much interest in the concept of functional classification and, in particular, the role of species traits in ecosystem functioning (Diaz *et al.* 2007; De Deyn *et al.* 2008). Supporting services are also strongly affected by geodiversity, namely the variety of rocks, minerals, fossils, landforms, sediments and soils in a place (Chapter 2). Not only does geodiversity support the provision of basic raw materials upon which supporting services, such as soil formation, are based, but it also influences biodiversity and its spatial variation at the habitat and landscape scale (Chapter 2).

Despite the above, our understanding of the mechanisms by which ecological interactions influence ecosystem processes and the delivery of supporting services is limited, as is our knowledge of their contribution relative to abiotic factors as drivers of supporting services at the landscape scale (Bardgett & Wardle 2010). The picture is complicated further by the knowledge that the effects of ecological interactions on supporting services, such as plant production and nutrient cycling, are strongly context dependent. For example, a number of studies indicate that the effects of biodiversity on ecosystem processes in terrestrial and aquatic ecosystems vary depending on environmental context (Fridley 2002; Covich *et al.* 2004; Wardle & Zackrisson 2005). There are also indications that the influence of biotic

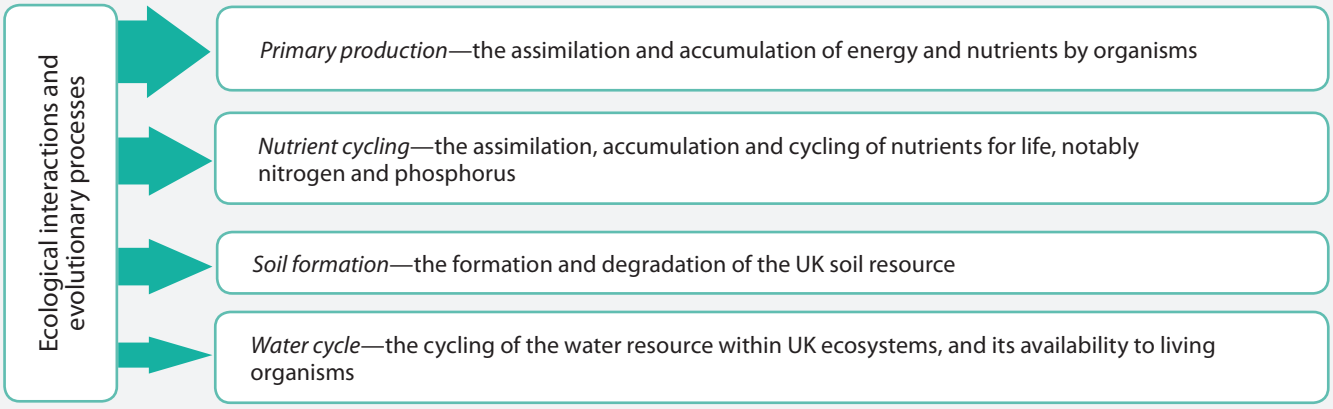

Figure 13.1 The supporting services of primary production, nutrient cycling, soil formation and water cycling, which are all underpinned by ecological interactions to differing degrees. The size of the arrows represents the extent that each supporting service is regulated by biotic/ecological interactions.

drivers (e.g. herbivory) on ecosystem processes varies along environmental gradients of climate, nutrient availability and topography (Olff *et al.* 2002; Bardgett & Wardle 2003; Anser *et al.* 2009). Given that such biological interactions and their relationship with the abiotic environment underpin the delivery of supporting services (**Figure 13.1**), we consider them as an integral part of our evaluation.

In this chapter, we provide an overview of the trends, past, present and future, which are evident in supporting services provided by the broad habitats of the UK, and consider what the main drivers are for these trends. We also evaluate what is known about the consequences of these trends and consider options for the sustainable management of supporting services in the future. Finally, we identify knowledge gaps regarding the delivery of supporting services in the UK.

13.2 Soil Formation

13.2.1 Condition, Status and Trends

Soils are formed by the weathering of rocks and minerals and the accumulation of organic materials, which often takes hundreds to thousands of years (**Box 13.1**). Soil formation is a continuous process and its speed and nature is affected by several factors (Jenny 1941) including the parent material, climate, topography, biota (including plants, animals and microorganisms) and land management. Rates of soil formation for mineral soils vary greatly, but they typically lay in the range 0.04–0.08 mm/yr; this would create soil at a rate of less than 1 cm per century (EASAC 2009). The loss of soil by erosion is also a natural process, but it can be very rapid and accelerated by land use. Moreover, rates of soil loss can be much greater than formation; hence, soils should be treated as a non-renewable resource. Data for rates of soil loss by erosion in the UK are few, although estimates of 0.02–1.27 tonnes/hectare/year for mineral soils, and as much as 10 t/ha/yr for tillage erosion from arable fields have been reported (Verheijen *et al.* 2009). Based on global literature, Wilkinson & McElroy (2007) estimated an average natural erosion rate of 0.4 t/ha/yr over 542 million years of the Phanerozoic, with a peak of 1.4 t/ha/yr in the Tertiary period.

A combination of soil processes and soil-forming factors is responsible for the formation of different soil types. On well-drained soils, humus formation, leaching, weathering, clay migration and clay alteration lead to the formation of podzols. On poorly drained soils, the mobilisation and precipitation of iron gives rise to gley soil; whereas low rates of organic matter decomposition, caused by waterlogging and low temperatures, leads to the formation of peats. In the UK, most areas have greater precipitation than evaporation and, as a consequence, leaching is a dominant soil process. However, slower processes of weathering are also of importance, as a primary source of essential elements (e.g. trace elements and base cations) for plants and soil organisms whose actions (mixing, dissolution, physical disruption) promote further weathering. For example, biological weathering occurs via roots which penetrate and crack open rocks, and via mycorrhizal fungi which form extensive hyphal networks in soil and so play a significant role in weathering minerals. The release of base cations from chemical weathering is also of importance because it replenishes those which are lost through leaching or biological uptake. Soil is also the primary natural source of essential trace elements in animals and man, although trace element concentrations depend strongly upon soil parent material, being lowest in soils derived from acid igneous rocks and sands. Hence, in soils derived from sedimentary and base rich rocks commonly found in England and Wales, approximately 5% of agricultural soils have low trace element status compared to 30% of agricultural soils in Scotland where soils are derived from igneous and metamorphic rocks and sands (Sinclair & Edwards 2008).

The interaction of so many soil-forming factors, which vary simultaneously in space and time, means that the UK has many diverse soil types (**Box 13.2**; **Figure 13.2**). Also, the combination of the maritime climate of the UK (with cool temperatures and relatively wet conditions) and the predominantly hard rock resistant to weathering, has given rise to soils that are much younger, contain more organic carbon (C) and experience more leaching and poorer natural drainage, than soils in eastern and southern Europe. This pedological youthfulness is due to the presence of ice sheets and periglacial conditions that only retreated from most of Britain 10,000–15,000 years ago. There is a marked contrast across Britain between the mineral soil types, which primarily form in the lowlands, and the soils in the uplands which have surface horizons rich in organic matter. The balance in terms of provisioning economically valued ecosystem services is often juxtaposed to important regulating services that have uncertain or no market value. For example, the uplands harbour most of the UK's organic soils which store most of the nation's carbon, compared to lowland areas where mineral soils of much lower organic

Box 13.1 Definition of Soil

Soil forms a thin mantle over the Earth's surface and acts as the interface between the atmosphere and lithosphere, the outermost shell of the Earth. Soil is a biologically active and complex mixture of weathered minerals, organic and inorganic compounds, living organisms, air and water, which provides the foundation for life in terrestrial ecosystems. Soil is not merely the sum of these parts; rather, it is a product of interactions between these components, and the formation of soil is dependent on these interactions. The rate at which these processes occur typically decreases with depth, high latitudes and altitudes. The depth at which these processes cease or become insignificant varies, but in mid–latitudes, it is generally between 1–2 m below the ground surface.

Within any landscape there is an incredible range of soils, resulting from almost infinite variation in the soil-forming factors of parent material, climate, biota, relief and time (Jenny 1941). These are highly interactive, all playing a part in the development of a particular soil. A specific combination of these factors leads to the development of a unique soil type, with a relatively predictable series of horizons (layers) that constitute the soil profile. Of most relevance for ecosystem service delivery, are those horizons that are at, or close to, the soil surface; this is where most microbes and animals live, and where most root growth and nutrient recycling occurs. These horizons are referred to as the surface organic O horizon, derived from decomposing plant and animal material, and the uppermost A horizon, composed largely of mineral material, but also intimately mixed with organic matter derived from the surface O horizon.

Box 13.2 Not one but many UK soils

Soils in the lowlands are predominantly **luvisols** (illuvial accumulation of clay), **cambisols** (weathering) and **gleysols** (redox gradient in soil profile, translocated iron and manganese compounds) with subtle differences between different parts of the UK (FAO-IUSS-ISRIC 2006; **Figure 13.2**). In Northern Ireland and the western part of the Midland Valley of Scotland, gleysols predominate, whereas over much of England luvisols are the most common soil types distinguished. In eastern Scotland and Wales, cambisols are most common. In the uplands, peat soils (**histosols**) (permanent anaerobic conditions) and gleysols predominate, with the exception of central and eastern Scotland where **podzols** (leaching, weathering, clay and iron migration) occur. Many of the gleysols and podzols have histic surface horizons due to low evapotranspiration rates causing organic matter build-up at soil surface. **Leptosols** (shallow soils suggesting very slow processes) are the dominant soils on the highest mountains and are extensive due to the large areas of bare rock in these environments and very shallow drift. They are also found on calcareous soils in southern and eastern England. **Fluvisols** (formed by depositional processes) are found associated with rivers and estuaries**.** This diversity of soils, and associated geology, determines the balance of soil-related functions in the UK landscape.

carbon content support most of the UK's agricultural production.

Data on the status of soils in the UK is generally lacking and we know little about trends with respect to future drivers. Two highly relevant issues for ecosystem service delivery are: a) the formation and loss of peat soils, largely due to their great importance as a carbon sink; and b) the loss of soil to urbanisation, which is considered in Section 13.2.2 of this chapter and in Chapter 10.

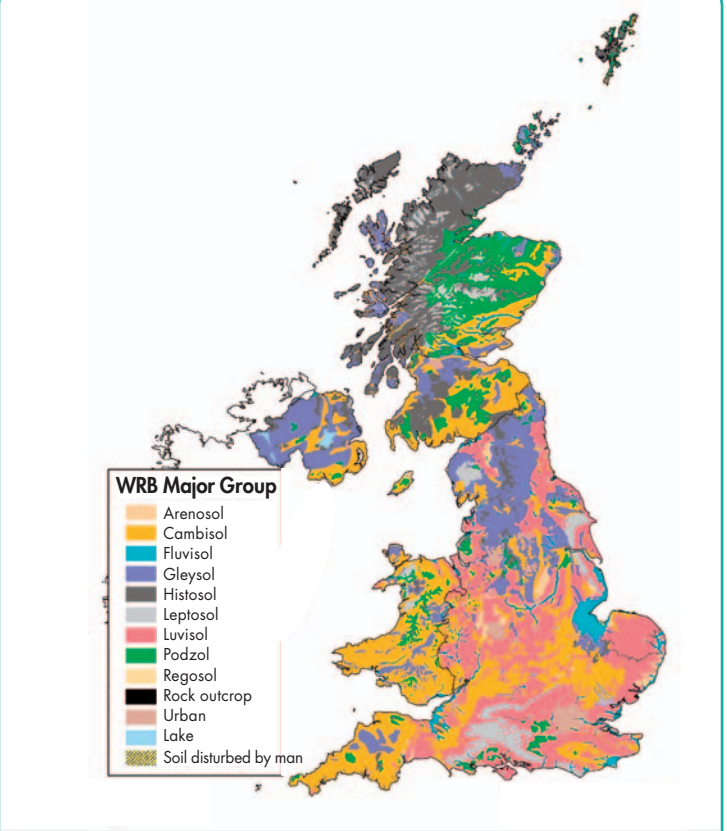

Figure 13.2 Principal Soil Types in the UK according to the World Reference Base for Soil Resources (FAO-IUSS-ISRIC 2006). Source: EU 1:1000000 soil map (European Soil Database (v 2.0), European Soil Bureau Network and the European Commission, EUR 19945 EN, March 2004).

Peatlands are a distinctive ecosystem associated with deep peat soils, which began to form in the UK 3,000–9,000 years ago following the retreat of the last ice sheet and the subsequent cool and wet period. Peat soils have developed to become over 10 m deep in places, although the mean depth, at least in Scotland, is 2 m (Chapman *et al.* 2009). The depth of peat soils is estimated to increase at a rate of 0.8 mm/yr in actively growing bogs of good habitat status, which is equivalent to a carbon accumulation rate of 0.5 tonnes of carbon/hectare/year. However, the average is probably closer to 0.1 tC/ha/yr given that many peatlands are not of good status (Joosten & Clarke 2002). The total peatland carbon stocks in Scotland and Wales have been estimated to be 1,620 megatonnes of carbon (MtC) and 116 MtC, respectively (Chapman *et al.* 2009; Smith *et al.* 2007), whereas in England and Northern Ireland C stocks have been estimated at 296 MtC and 90 MtC, respectively (Bradley *et al.* 2005) but only to a depth of 1m. This estimate therefore does not include the unknown stock of carbon below 1m depth for England and Northern Ireland. The current rate of carbon fixation for all UK peatland is difficult to determine, but a conservative estimate based upon the above figures gives 0.27 MtC/yr; others have estimated 0.7 MtC/yr (Cannell *et al.* 1999), assuming that the peatlands are in good condition. In practice, many areas have been subjected to draining, agriculture and peat cutting, and losses have been estimated at 1 MtC/yr (Cannell *et al.* 1999). Other peatlands have been subjected to afforestation, burning, overgrazing, atmospheric pollution and extensive erosion (**Figure 13.3**). Much of the carbon is lost as carbon dioxide to the atmosphere, but a significant proportion is lost to water as dissolved organic carbon (DOC) and particulate organic matter (POM), the ultimate fate of which is largely unknown. Total UK fluvial losses of carbon in the form of DOC and POM have been estimated at 1 MtC/yr (Cannell *et al.* 1999) with much of this arising from peat soils and having significant consequences for water quality (Chapter 14).

Figure 13.3 Peat erosion within highland landscape of Ladder Hills, Scotland. *Photo © A.J. Nolan / Macaulay Land Use Research Institute.*

13.2.2 Drivers of Change

In the UK, humans have been managing and interacting with soils for over 5,000 years. This long-term relationship has contributed to the diversity of the soils found here and the uniqueness of the British landscape. Furthermore, because UK soils are well-fed by rainwater and well-endowed with organic matter, they are often considered to be more fertile and resilient than soils elsewhere in Europe, especially southern Europe, where erosion is more intense and water is in short supply. Indeed, recent status reports conclude that UK soils are generally in good health, although there are significant potential threats such as carbon loss, erosion, compaction and contamination (Environment Agency 2004; Towers *et al.* 2006). Here, we identify the main drivers of change in soil formation and associated threats to UK soils.

13.2.2.1 Land use

Land use is a major driver of change in soil formation through soil erosion and the accumulation or loss of soil organic matter. Rates of soil formation are slow, yet anthropogenic interventions and management can significantly change the properties of surface soil in only a few years. In the last 50 years, UK agricultural soils have been subject to significant change, particularly through increased mechanisation and use of agrochemicals and synthetic fertilisers, and through the introduction of new crop varieties. Increased frequency and depth of tillage, reductions in land under forage crops, and continuous use of grain cereals have all led to major concerns about declines in soil organic matter, with significant losses of top soil carbon in some circumstances (Johnson *et al.* 2009). For example, the ploughing, liming and fertilising of podzols in lowland England over the last 30 years, has transformed them into brown podzolic soils of lower soil surface carbon content. Changes may also be occurring in the uplands; the growth of birch on heather moor can change the soil from a carbon sequestering peaty podzol to a brown podzolic in less than 40 years (Mitchell *et al.* 2007). In addition, increased grazing pressure and associated land improvement in the uplands (e.g. liming and fertiliser application) over the last few decades have led to the widespread conversion of semi-natural grassland on brown podzlic soils into species-poor improved grassland with brown earths of lower carbon content (Bardgett *et al.* 2001; Grayston *et al.* 2004).

Over the next 50 years, potential changes including the introduction of biofuel crops, increased recycling of organic wastes and by-products to land, and reductions in fertiliser application in response to higher oil and energy costs, could have significant impacts on UK soils. Moreover, the UK intends to increase its amount of forested land. If significant new planting of trees on agricultural soils occurs, this could greatly influence soil formation because trees root deeper and accumulate litter and organic matter at the surface, with likely consequences for physical properties and nutrient cycling (Carroll 2004).

13.2.2.2 Climate change

The changing climate is one of the greatest threats to UK soils, although its impact will vary with soil type and other pressures placed on soils. Major changes or damage to soil due to climate change is anticipated in the coastal zones, such as the machairs (fertile low-lying grassy plains found on some of the north-west coastlines of Ireland and Scotland) and Links soils of Scotland which are directly under threat from sea-level rises (Angus *et al.* 2010; Rennie & Hansom in press). The formation of montane soils, where freezing cycles are an important soil-forming process, is also likely to diminish if the climate warms. Current trends indicate that winters will become warmer and wetter, and summers drier. These changes will affect rates of soil weathering and could increase both water and wind erosion. Warming of organic soils is likely to exacerbate loss of soil carbon through accelerated decomposition of organic matter (Dorrepaal *et al.* 2009). Increased drought and drying in wetlands and peatlands, which will lower the water table and introduce oxygen into previously anaerobic soil, will create more favourable conditions for microbial activity and, therefore, cause carbon loss (Freeman *et al.* 2004). Also, long-term climate change experiments in upland heathland have demonstrated the potential for cumulative effects of repeated summer drought on soil carbon dioxide flux, which increased year on year to be 40% greater than the control after six years (Sowerby *et al.* 2008). These findings demonstrate the risk of peatlands and wetlands being degraded under climate change, with implications for the global carbon cycle.

Climate change may also alter the magnitude, frequency, rate and nature of soil-forming processes, resulting in enhanced rates of process activity, including less recovery time between extreme events. Some of the most dramatic impacts are likely to be in coastal areas where the effects of climate change will be compounded by sea-level rise (Rennie & Hansom in press). This, in turn, will impact on coastal landforms, habitats and soils, resulting in issues such as enhanced coastal retreat and steepening, coastal squeeze (where landward migration of landforms and habitats is impeded) and enhanced landslide activity on susceptible coasts (Angus *et al.* 2010). In some cases, rates of change may exceed those seen in the last 7,000 years or more (Orford & Pethick 2006; Rennie & Hansom in press), leading to widespread reorganisation and disturbance of coastal landforms, habitats and soils (Angus *et al.* 2010; **Figure 13.4**).

Figure 13.4 Variation in coastal morphology at Culbin Sands Moray. *Photo © P. & M. Macdonald/Scottish Natural Heritage.*

While The Intergovernmental Panel on Climate Change's (IPCC 2007) projections of future sea-level rise cover a wide range, they are considered to be conservative in the light of observational data and because the dynamic responses of ice sheets have not been accounted for (Pfeffer *et al.* 2008; Rahmstorf 2007). Sea-level rise will not only influence the tidal immersion of coastal habitats, but also rates of erosion, sediment transport and accretion. Sea-level rise will also increase brackish water encroachment on lowland coastal areas, leading to localised risk of soil, habitat and groundwater salinisation (Orford & Pethick 2006). This will impact on several coastal Sites of Special Scientific Interest (SSSIs) and other protected habitats (Angus *et al.* 2010). Climate change scenarios suggest that some land-forming processes that are also hazardous, such as coastal flooding and erosion, flash floods and landslides, are likely to occur more frequently (Orr *et al.* 2008). The response to hazards often results in expensive site-by-site geotechnical solutions, but many of these approaches are unsustainable and may exacerbate or transfer the problem elsewhere in the catchment or along the coast, with additional impacts (Prosser *et al.* 2010).

13.2.2.3 Urbanisation

When land is used for built infrastructures, such as houses, roads and industrial structures, the loss of soil as a resource in itself is complete and, in most cases, irreversible within our lifetime. Often, urban expansion takes over good agricultural land because population centres were historically established in close proximity to food supplies. Data on the conversion of land primarily in agricultural use to urban development (**Figure 13.5**) shows that, in England, the net conversion of agricultural land to urban use amounted to between 5,000 and 6,000 ha/yr between 1989 and 2003 although it has fallen markedly in more recent years, whereas in Scotland it was 1,200 ha/yr between 1995 and 2002 (Towers *et al.* 2006). Given the huge difference in population between the two countries and, by implication, the demand for new housing, roads and development projects, the loss in Scotland is surprisingly large, although there may be differences in the definition of 'sealed land'. In Northern Ireland, a net conversion rate of agricultural land to urban (both urban and suburban) of 1,000 ha/yr was found for the period 1970 to 1990 (Cannell *et al.* 1999). One target for development in England is that 60% of new housing should be built on previously developed and/or abandoned industrial land ('Brownfield sites'); this has been attained every year since 2000. In the Countryside Survey, the area of 'Built-up areas and Gardens Broad Habitat' in Wales was estimated to have increased by 14,700 ha between 1998 and 2007 (Smart *et al.* 2009).

13.2.3 Consequences of Change

There are multiple consequences of loss of soil, or change in soil formation rate, for all other supporting services, and the regulating and provisioning services that they underpin. Even partial loss or reorganisation of soil surface horizons has significant consequences for the delivery of other ecosystem services. For example, a total or partial loss of soil represents a loss of nutrient supply capacity with knock-on consequences for primary production and the provisioning and regulation of the quantity and quality of water. Also, total or partial loss of soil can reduce our capacity to produce food and fibre, which is dependent on the maintenance of soil fertility. For example, if the loss of 1,200 ha/yr in Scotland continued for seven years, and all

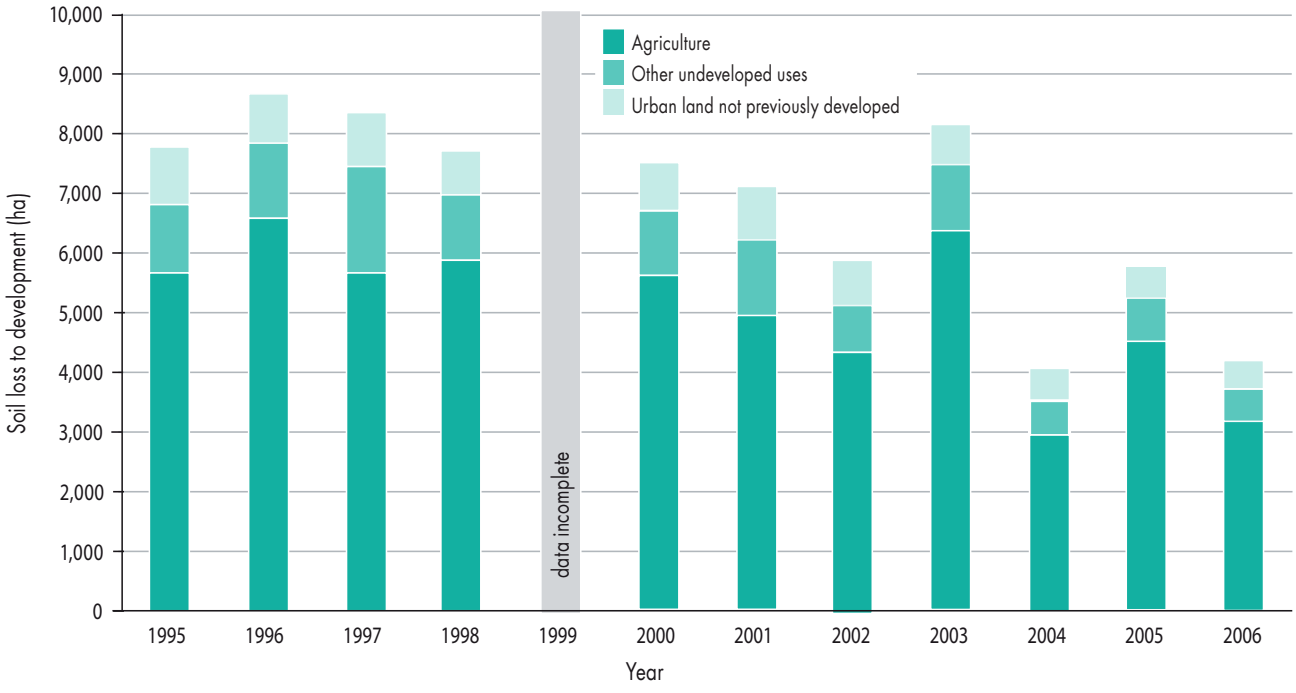

Figure 13.5 Soil loss to development in England, 1994 to 2006. Source: data from the Department of Local Communities and Government (DLCG 2010).

this land was good agricultural land, it would be equivalent to losing 1.5%, 1.4% and 1.25% of the national production of spring barley, wheat and potatoes respectively. Soil loss affects the regulation of soil fertility and climate because soil acts as a major sink for carbon and source for greenhouse gases (Chapter 14). It has been estimated that around 2.2 million tonnes of top soil is eroded annually in England and Wales, affecting agricultural production and nutrient availability (Quinton *et al.* 2010), and also water quality (Environment Agency 2004), with the total annual costs being around £45 million, including £9 million in lost production (Environment Agency 2007). This also causes a loss of soil biodiversity and the ecosystem services that it underpins, which has been estimated at approximately $3 trillion worldwide (Pimentel *et al.* 1997) excluding the economic benefits of primary production.

The depth of soil to the parent material represents the total amount of soil available for performing functions and ecosystem services, and the soil column is differentiated into soil horizons whose individual properties and characteristics control many of the environmental interactions. Pimentel and Pimentel (2000) suggested a minimal soil depth of 15 cm was required for agricultural production. The depth of some soils can be as low as 10 cm, while some peat soils are over 10 m deep. However, depth alone is not an indicator of the status or age of soil; the rate of soil formation can be significantly reduced under harsh climatic conditions, such as those found on mountains, where soils may show well-differentiated and fully functional soil horizons of only a few centimetres in depth.

Using depth and bulk density values it is possible to estimate stocks in various currencies. The total mass of soil in the UK is 200,000 Mt, which contains 10,000 Mt of carbon, 440 Mt of nitrogen and 120 Mt of phosphorus (**Table 13.1**). It can also be estimated that, when fully saturated, UK soils store 130 billion m^3 of water (**Table 13.1**). To put this into perspective, the amount of water stored in soil in Scotland alone (42 billion m^3) is more than all the fresh water found in Scottish lochs. Given the importance of soil carbon for other ecosystem services, and especially climate regulation (Chapter 14), using organic matter accumulation as a proxy for soil formation is a useful way of valuing soil formation in terms of the full social costs as there are established prices structures in different markets. Of the 10 billion tonnes of carbon stored in UK soils (Smith *et al.* 2007) most is found in organic soils in semi-natural montane, moor and woodland habitats (including peat soils), and carbon is lower under enclosed agricultural land and settlements (Bradley *et al.* 2005).

Table 13.1 Estimates of total mass of soil, soil carbon (C), nitrogen (N), phosphorus (P), calcium (Ca), magnesium (Mg), potassium (K) and volume of water in saturated soil for the UK. Values were calculated for Scotland using the Scottish Knowledge and Information Base and extrapolated to UK by area.

	Total Mass in top 1 m (millions of tonnes)							Volume of water (billion m^3)
	Soil	C	N	P	Ca	Mg	K	
Scotland	64,000	3,000	142	40	45	9	3.5	42
Total for UK	200,000	10,000*	440	120	140	27	11	130

*Smith *et al.* (2007)

13.2.4 Options for Sustainable Management

There are several management options for enhancing soil formation and minimising soil loss. New soil strategies in the UK and EU (Defra 2009; European Commission 2002, 2006a and 2006b; Scottish Government 2009) recognise soil as a finite resource and the need to protect it from carbon loss, compaction and contamination, setting out measures for more sustainable management. The appropriate use of bulky organic fertilisers, such as manures and composts, will enhance the accumulation of soil organic matter. Similarly, different crop rotations with ley crops and the use of cover crops can increase organic matter and help to minimise erosion losses. In the arable sector, minimal tilling and seed-bed preparation help to minimise direct loss of soil from erosion. Certain crops which require intensive cultivation, such as potatoes, can be confined to relatively gentle slopes and contour ploughing, although this is not widely practiced. Inappropriate grazing pressure can also exacerbate soil loss, notably on vulnerable, highly organic soils. And forestry management now encourages much less destructive forms of site preparation prior to planting, minimising disturbance to soils.

Management strategies aimed at maintaining, or enhancing, the accumulation of soil organic matter can have multiple synergies for provisioning, supporting and regulating services (Lal 2008; Smith *et al.* 2008a; Woodward *et al.* 2009). For example, an increase in soil organic matter content can afford benefits for soil fertility such as improved soil structure and water-holding capacity, greater complexity and diversity of the soil food web, binding and transforming pollutants that might otherwise enter the food chain or water supplies, and increased storage and retention of nutrients and water (Lal 2008; Woodward *et al.* 2009). Increased soil organic matter accumulation can also have synergies with biodiversity conservation: planting of high-diversity mixtures of native grassland perennials on degraded, low organic matter content soils can yield advantages over monocultures in terms of productivity, reduced greenhouse gas emissions and carbon storage (Tilman *et al.* 2006), with additional benefits for wildlife conservation. Positive effects of plant diversity for soil carbon sequestration have also been reported in grasslands, although this has been attributed to the presence of particular plant functional groups, most notably legumes, rather than diversity *per se* (Fornara & Tilman 2008; Steinbeiss *et al.* 2008; De Deyn *et al.* 2009).

Soils take a long time to form, but a short time to degrade. This has led many to call for a long-term land use strategy that considers the appropriate time-scales of soil formation and accounts for the complex synergy between soil-forming processes. For example, planning policies used to guard against building on prime agricultural land, but this protection was lost during the 1980s. With food security a key issue and the realisation of soil loss rates to building, some soil strategies are suggesting this protection of prime land should be reinstated (Scottish Government 2009).

13.2.5 Knowledge Gaps

Despite an historical interest in the study of soil formation, our understanding of the mechanism and rates of soil formation and soil loss for key soil types is limited. There are few models of soil formation, in contrast to the many models available for predicting soil loss by erosion. Our understanding of erosion processes in organic soils and the role of biodiversity in soil formation is limited. However, there is increasing recognition that approaches based on plant functional diversity (i.e. the range, type and relative abundance of plant functional traits) offer potential for understanding the impact of biodiversity on supporting services, including soil formation and carbon accumulation (Díaz *et al.* 2007; De Deyn *et al.* 2008).

Soil is also an ecosystem in its own right and changing its formation process will impact on the habitat of soil biota, as well as interact with above-ground biodiversity. Therefore, the sensitivity and responses of habitats to climate change and sea-level rise will depend, in part, on how the underlying soil processes respond. Climate change will lead to changes in land use and land management practices as a result of changes in the suitability of land for agriculture, forestry and renewable energy production, along with indirect pressure from population growth (such as food policies) and displacement (planning development away from flood-prone areas). The consequences for soil formation of such increased pressures on the land are poorly understood.

Changes in the rate of accumulation or loss of organic matter in UK soils are of central importance for ecosystem service delivery, as discussed above. Two large-scale surveys which have monitored change in soil carbon have been completed, but they have found contrasting results: Bellamy *et al.* (2005) reported significant losses, whereas the Countryside Survey (2010) found no change. (The difference between the two monitoring schemes remains unresolved and is the subject of continuing studies.) As with most studies of soil carbon, both surveys were conducted on surface horizons, but future studies need to understand the distribution and dynamics of carbon in deeper soil horizons, including that below the plough layer (Neff *et al.* 2002). This is because significant quantities of carbon can be found in deep horizons and the soil processes that regulate the turnover of this carbon are quite different from those that operate in superficial layers. The depth of soil and its spatial variation is difficult to measure, but such an assessment of soil carbon stocks in relation to depth is of fundamental importance in determining the natural capital stock of our soil resource, the soil organic carbon and nutrients required to support ecosystem services, and the links to human well-being.

13.3 Nutrient Cycling

13.3.1 Condition, Status and Trends

Although data on nutrient cycling in different UK habitats is sparse, several national surveys have provided information on key trends, including the Countryside Survey (2010), the UK Environmental Change Network (ECN), which since 1992 has been monitoring soil chemistry at 12 locations throughout the UK as well as water quality in 16 lakes and 29 rivers, and the Review of Transboundary Air Pollution (RoTAP 2011), which has reported on the extent and impact of nitrogen (N) deposition across the UK. Here, we identify some of the main trends in nutrient cycling that have emerged from these surveys and other studies.

13.3.1.1 Nitrogen

The mineralisation of nitrogen—the process by which soil microbes break down organic nitrogen and convert it into inorganic forms—is of critical importance because, in many habitats, it determines the availability of nitrogen for primary production. There are major differences in amounts of nitrogen and rates of nitrogen mineralisation in top soils (0–15cm) across UK terrestrial habitats, which broadly relate to the total amount of organic matter present and whether soils have been agriculturally improved (Emmett *et al.* 2010; **Figure 13.6a,b**). For example, total amounts of nitrogen in topsoil of habitats that have had no fertiliser added, such as coniferous woodland and wetlands (approximately 4 tN/ha), is lower than in grassland soils, especially those which have been improved for intensive agriculture (approximately 6 tN/ha). On the other hand, rates of nitrogen mineralisation vary with organic matter content: arable soils, which contain relatively little organic matter, mineralise far more nitrogen per unit of organic matter than upland or woodland soils,

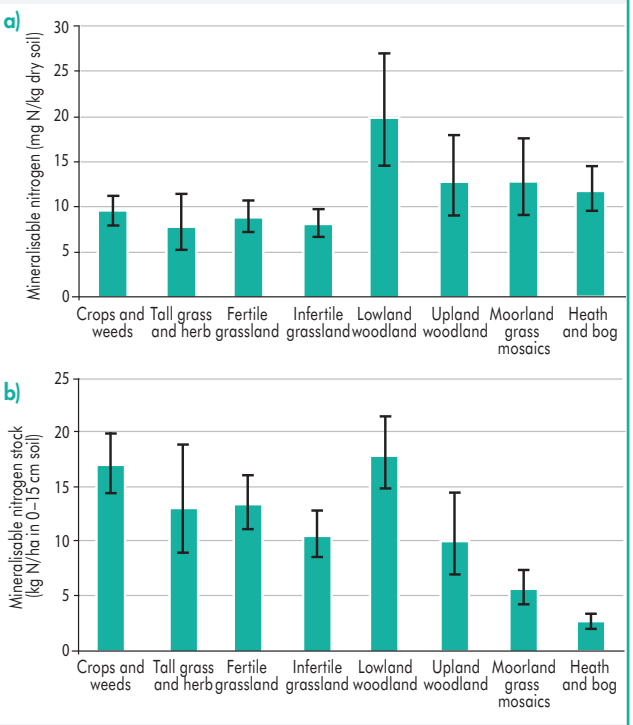

Figure 13.6 Mean (+/- one standard error) nitrogen stock and availability in soils (0–15 cm) in aggregate vegetation classes: a) mineralisable nitrogen concentration in soil organic matter (mg N/kg loss-on-ignition); and b) stock of total mineralisable nitrogen (kg N/ha). Source: reproduced from Emmett *et al.* (2010). Countryside Survey data owned by NERC – Centre for Ecology & Hydrology.

which contain relatively more organic matter (Emmett *et al.* 2010). As a consequence, the concentration of nitrate, which is highly mobile and readily lost from soil by leaching and denitrification, is greater in intensively managed grassland and arable soils (Emmett *et al.* 2010). In contrast, the concentration of dissolved organic nitrogen (DON) tends to be greater in soils of semi-natural and less intensively managed grassland than in improved grassland soils, and this DON is also used more readily by plants in these less fertile situations (Bardgett *et al.* 2003).

The most dramatic trend in nitrogen cycling over the last 50 years has been the enrichment of UK terrestrial habitats with nitrogen due to the application of fertiliser nitrogen in managed land and atmospheric nitrogen deposition in semi-natural systems. This enrichment of UK habitats has been detected through the analysis of national-scale changes in plant species composition between 1990 and 1998, which revealed apparent eutrophication effects across several common British vegetation types including infertile grasslands, moorland, upland woodlands and heath/bog (Smart *et al.* 2003; Smart *et al.* 2004). Although a UK decline in emissions of both oxidised (50%) and reduced nitrogen (24%) has been observed during the period 1984 to 2005, there is no evidence that this has led to a decline in nitrogen deposition (RoTAP 2011). As a consequence, semi-natural habitats in the UK are still subject to nitrogen deposition, often exceeding 25 kg N/ha/yr, and so, remain at risk of unintended nitrogen enrichment and damage. Little is known about the fate of atmospheric nitrogen and the capacity of UK soils to retain this nitrogen, but in habitats where primary production is limited by nitrogen availability it is likely that anthropogenic nitrogen will accumulate in soil organic matter (Nadelhoffer *et al.* 1999; Phoenix *et al.* 2004; RoTAP 2011). This can slow down, or even halt, deleterious effects of nitrogen deposition (Aber *et al.* 1989). Despite this ongoing deposition of nitrogen, a recent synthesis of data from 12 terrestrial sites in the ECN between 1993 and 2007 (Morecroft *et al.* 2009) found no consistent temporal patterns in concentrations of inorganic nitrogen (ammonium or nitrate) in soil solution suggesting that nitrogen is being retained within the terrestrial system or lost as nitrogen gas. The finding of the Countryside Survey (2010) of an increase in the carbon:nitrogen ratio of topsoil across most UK terrestrial habitats, with the exception of arable habitats, supports the idea that nitrogen enrichment has increased plant production and, therefore, the fixation of carbon and its transfer to soil (Emmett *et al.* 2010).

Data on nitrogen cycling in aquatic habitats is mainly focused on water quality, which is considered in detail in Chapter 14. However, of high relevance for aquatic nitrogen cycling is the trend of declining nitrate (NO_3) concentrations in rivers over the last 10 years; although a large proportion of river length in the east of England still exceeds 30 mg NO_3/l, this proportion has declined since 2000 (from 90% to 60% in the Anglian region and from 50% to 40% in the Midlands) (Environment Agency 2008). In contrast, the length of rivers in the west of England exceeding 30 mg NO_3/l has remained steady at 10–20% since the year 2000 (**Figure 13.7a**; Environment Agency 2008). Traditionally, phosphorus (P) is considered to be the main nutrient limiting primary production in freshwater habitats (Schindler 1977), whereas in coastal and marine habitats it is thought that nitrogen is the most limiting nutrient (Howarth 1988). Therefore, nitrogen is the element, which in excess, commonly leads to eutrophication (Howarth & Marino 2006). However, recent work has questioned these generalisations, indicating an equivalence in nitrogen and phosphorus limitation in lakes (Elser *et al.* 1990) and streams (Francoeur 2001), and frequent phosphorus limitation in the oceans (Downing *et al.* 1999). Via a large-scale meta-analysis of experimental enrichments, it has also been shown that phosphorus limitation is equally strong across terrestrial, freshwater and marine habitats, and that nitrogen and phosphorus limitation are equivalent within both terrestrial and freshwater systems (Elser *et al.* 2007). Although not technically eutrophic, the Irish Sea has been found to contain raised levels of anthropogenic-nitrogen (Gowen *et al.* 2008) and there was a trend of increasing nitrate concentrations during the period 1960 to 1980, followed by a decline in the 1990s (Evans *et al.* 2003). However, data on water column nutrient concentrations measured at an English Channel site from 1930 to 1987 shows a wide range in the nitrate:phosphate ratio and, for a significant number of years, mid-summer values of phosphate increased for short periods of time whilst nitrate concentrations remained low (Jordan & Joint 1998).

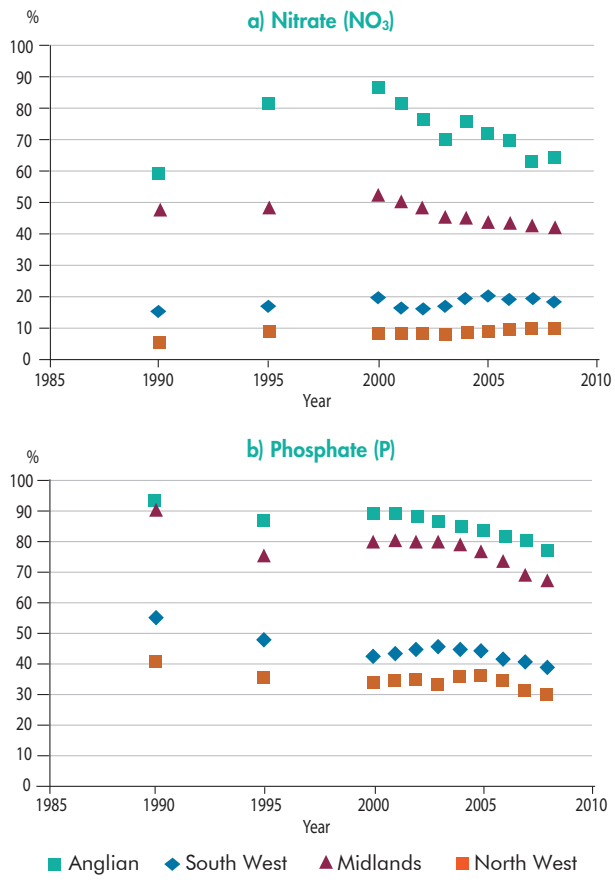

Figure 13.7 Percentage lengths of rivers in different regions of England containing: a) 30 mg NO_3/l or more; and b) 0.1 mg P/l or more. Source: Environment Agency (2008).

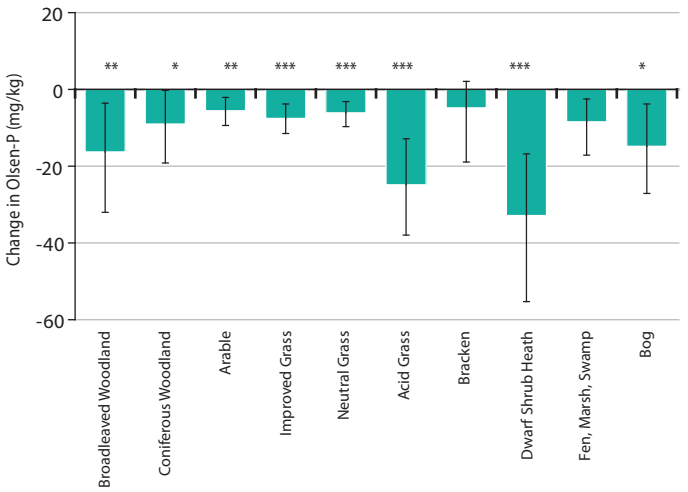

Figure 13.8 Change in mean Olsen-P concentration within Broad Habitats in GB between 1998 and 2007. Error bars are 95% confidence intervals, and *, **, *** refer to significant changes at the p<0.05, p<0.01 and p<0.001 level. Source: reproduced from Emmett *et al.* (2010). Countryside Survey data owned by NERC – Centre for Ecology & Hydrology.

13.3.1.2 Phosphorus

The Millennium Ecosystem Assessment (Lavelle *et al.* 2005) stated that phosphorus was accumulating in global ecosystems at a rate of between three and 10 times greater than that preceding the industrial era, largely due to agricultural activity. Despite this, the UK Countryside Survey (Emmett *et al.* 2010) found that Olsen-P (a measure of the amount of soil phosphorus available to plants) has unexpectedly decreased across UK habitats between 1998 and 2007 (**Figure 13.8**). The average decline across all UK habitats is reported to be about 25% from 42.8 mg P/kg soil in 1998 to 31.8 mg P/kg soil in 2007, whereas in some habitats, such as heathland and acid grassland, Olsen-P has halved. The reasons for this indication of a decline in soil Olsen-P are not understood and require further investigation. Applications of phosphorus fertiliser in the UK have declined since 1998, probably because its price has gone up substantially (Cordell *et al.* 2009); the amount of phosphorus applied to land in the UK fell by more than one third between the early 1990s and 2008 (BSFP 2009). During the same period, there was a steady decline in livestock numbers (Chapter 15) and applications of phosphorus to grassland almost halved during the period 2004 to 2008, and fell by about one third on arable land during the same period (British Survey of Fertiliser Practice 2009). Long-term experimental studies on cropping systems also suggest that Olsen-P is relatively sensitive to changes in fertiliser phosphorus inputs under continuous cropping (Aulakh *et al.* 2007; Messiga *et al.* 2010), with sharp declines in Olsen-P concentration over 5–10 years observed in treatments where phosphorus additions have been discontinued.

Whilst these changes in inputs and agricultural practices may explain declines in intensively managed land, the declines in semi-natural habitats are not readily explained and require further investigation before they can be confidently accepted. Moreover, caution is needed because the Olsen-P extraction technique is better suited to the analysis of calcareous and limed soils rather than the acid ones in which the largest changes have been detected. Although there is no evidence from the quality control procedures used in the Countryside Survey to suggest that the performance of the analysis has changed between the surveys (Emmett *et al.* 2010), the reported declines in phosphorus availability are larger and faster than would be expected from what is known of the chemistry of phosphate in soil. Therefore, further research is needed to support the finding. Overall, the results highlight our lack of knowledge about the phosphorus cycle and controls over transfers and mobility in many UK terrestrial habitats. They indicate that further research is required to better understand the factors that regulate the availability of this important nutrient across UK habitats.

As highlighted above, freshwater habitats are generally considered to be phosphorus limited (Schindler 1977) hence, their enrichment with phosphorus promotes primary production. However, the Environment Agency (2008) (**Figure 13.7b**) data suggests a trend of decreasing phosphate concentrations in the highly loaded rivers in arable regions of England and Wales between 1990 and 2008, and a less steep decline in phosphorus loadings from a lower baseline in high rainfall, livestock-rearing areas in the west of the country. The decrease in river water phosphorus is consistent with the decrease found in soil by the Countryside Survey (Emmett *et al.* 2010; **Table 13.1**), but it is unclear whether the two are causally linked. Integrated assessment of the soil, headwater and plant community data in the Countryside Survey has identified a clear link between changes in more woody and tall vegetation along riverbanks and lower phosphorus concentrations (Smart *et al.* 2010). The Scottish Environment Protection Agency (SEPA 2009) also found a trend of declining phosphorus in Scottish rivers, but this was attributed to better sewage control (Chapter 14). In the Irish Sea, there is evidence of a general trend, which parallels that for nitrate, of increased phosphate concentrations during the period 1960 to 1980, followed by a decline in the 1990s (Evans *et al.* 2003). In addition, the decline in marine phosphate load in the late 1980s and early 1990s paralleled that observed in rivers that feed into the Irish Sea (Evans *et al.* 2003).

13.3.1.3 Soil acidity and trace elements

The Countryside Survey revealed that the average pH of surface soils (0–15 cm) has increased across UK habitats between 1998 and 2007 (Emmett *et al.* 2010), continuing a trend observed between 1978 and 1998 (Countryside Survey 2000) (**Figure 13.9**). Although the pH of acidic, organic-rich soils and their associated habitats increased across the UK between 1978 and 1998, this trend has recently slowed (Emmett *et al.* 2010). An increase in soil pH has also been detected across terrestrial sites of the ECN, where soil solution pH at depth (i.e. at the bottom of the sub-surface B horizon) has been shown to have increased by about 4% between 1993 and 2007 at most sites (Morecroft *et al.* 2009; **Figure 13.10b**). This trend of increasing soil pH has also been found in other soil monitoring programmes (Kirk *et al.* 2010; Kirby *et al.* 2010; RoTAP 2011). Furthermore,

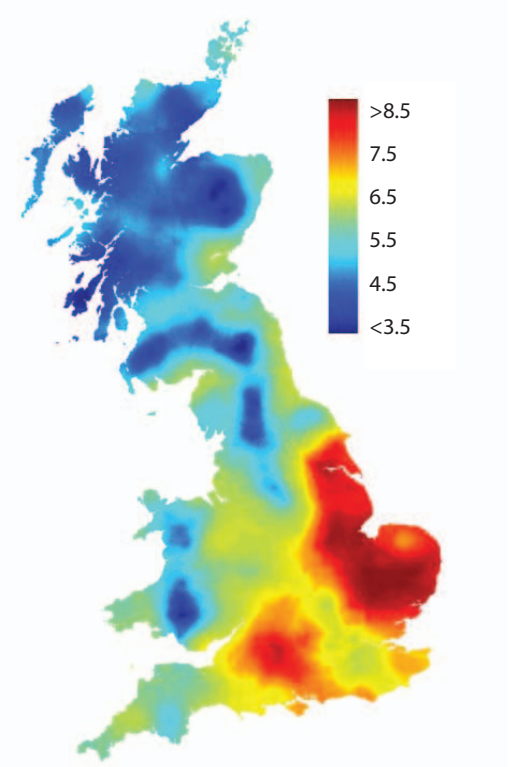

Figure 13.9 Map of soil (0–15 cm) pH in 2007 using ordinary Kriging (pH units). Source: reproduced from Emmett *et al.* (2010). Countryside Survey © Database Right/ Copyright NERC–Centre for Ecology & Hydrology. All rights reserved.

significant declines in soil solution sulphate concentrations were detected in surface soils (i.e. in the surface A horizon) of some of the upland sites of the ECN during this period, although no trends in this measure were found at the majority of sites. These trends appear to be related to the substantial decline in sulphate deposition that was detected at all sites over the monitoring period and which, in most cases, was accompanied by an increase in rainfall pH (**Figure 13.10a**) reflecting the introduction of emission controls in the mid-1970s (Morecroft *et al.* 2009). The widespread decline in soil acidity is associated with a reduction in habitats exceeding critical loads of acidity from 71% in 1996 to 1998, to 58% in 2004 to 2006; this is projected to decline to 40% by 2020 (RoTAP 2011).

Many trace elements are essential for primary production; their concentrations are related primarily to parent material, but their bioavailability is also affected by soil pH. In general, most trace elements become more available to plants and microbes in neutral or slightly acid soils, although molybdenum, for example, becomes more available in alkaline soils. In the UK, boron deficiency tends to occur on lighter soils where it is readily lost due to leaching; copper can be deficient in acid or alkaline soils, such as the shallow chalks of south-east England or acid peat and heathlands; and, similarly, zinc is deficient in high pH soils. With acid sulphate deposition, sulphur nutrition in UK crops was adequate. However, the recent decline in sulphur deposition, and the development of high yielding crop varieties, has meant that it is now necessary to add sulphur fertiliser to UK soils (Zhao *et al.* 2002). The uptake of selenium is also influenced by sulphur: plant uptake

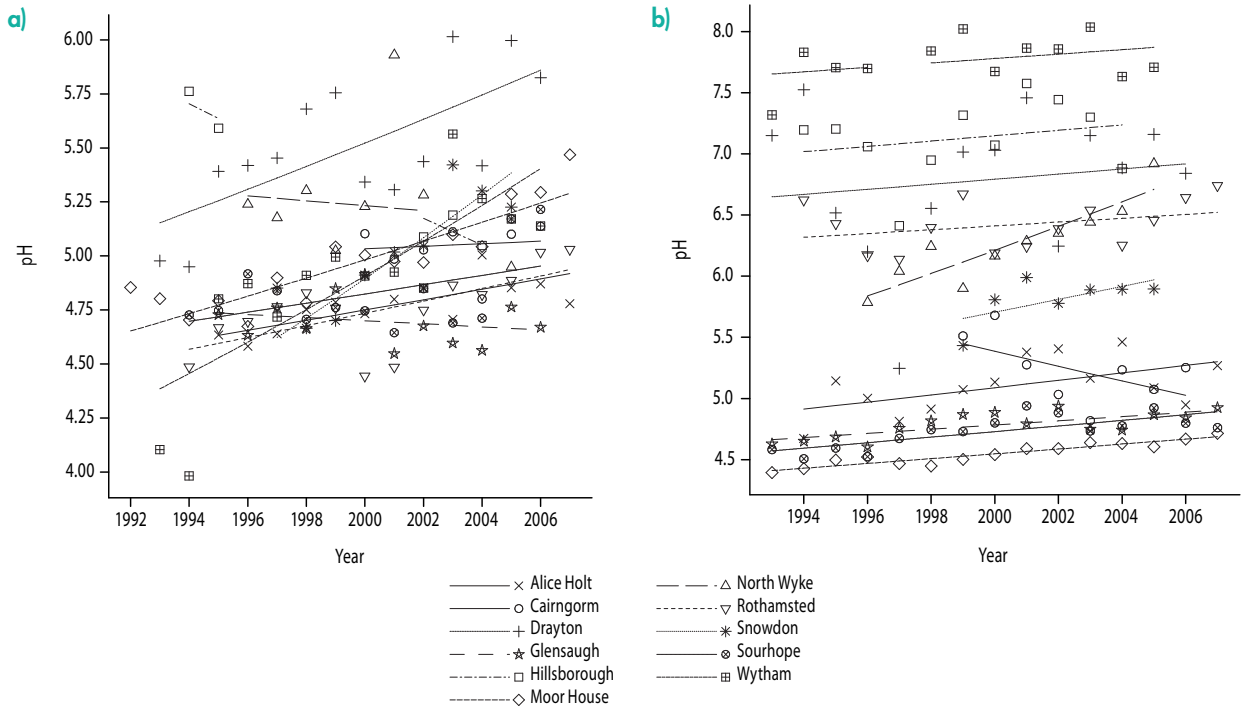

Figure 13.10 Increases in pH in both surface and subsurface soils across 12 terrestrial sites of the UK Environmental Change Network between 1993 and 2007. Figures show rise in pH across sites in a) rainfall (mean monthly); and b)subsoil (mean annual). Source: Morecroft *et al.* (2009). Copyright (2009), reproduced with permission from Elsevier.

of selenium is mediated by the same transporters as for sulphate, so as concentrations of sulphur fall, plant uptake of selenium increases (Fan *et al.* 2008; **Figure 13.11**). As a consequence, UK crops now contain adequate selenium for human nutrition, but livestock farmers generally supplement diets with selenium (Combs Jr. 2001).

13.3.2 Drivers of Change

13.3.2.1 Land use

Land use and agriculture, in particular, has exerted considerable influence on nutrient cycling across all UK terrestrial habitats, although the mechanisms involved are complex and poorly understood. Changes in land use can influence nutrient cycling in many ways, both direct (e.g. via fertiliser application, liming and cultivation, and the transport of nutrients from land to water) and indirect (e.g. via plant-soil feedbacks that influence rates of nutrient cycling and plant production). Indeed, increased grazing pressure across upland habitats has caused marked shifts from fungal- to bacterial-dominated microbial communities in soils, which are typically associated with enhanced rates of nutrient cycling and increased plant nutrient availability (Bardgett *et al.* 2001). In contrast, extensive management of upland habitats (e.g. via reductions or removal of grazing, or the cessation of fertiliser use) promotes fungal-dominated soil communities with 'slow' and highly conservative cycling of nutrients (Bardgett & McAlister 1999; Smith *et al.* 2008b), which contributes to reduced plant production and nutrient loss from soil (Van der Heijden *et al.* 2008). Free-range grazing, which is widespread in UK upland habitats, can also redistribute substantial amounts of nutrients within habitats because grazing animals feed and defecate in different places. For example, by concentrating phosphorus (and other nutrients) in small areas where they rest and ruminate, grazers can promote significant phosphorus loss from habitats, therefore, causing a gradual decline in their productivity (Jewell *et al.* 2007). Land use can also act as a major driver of changes in nutrient cycling in aquatic habitats, largely via the transfer of excess nutrients from land to freshwater, estuaries and the sea. Also, excess nutrient loading through river-runoff can act as a threat, although, as identified in Chapter 12, this is diminishing.

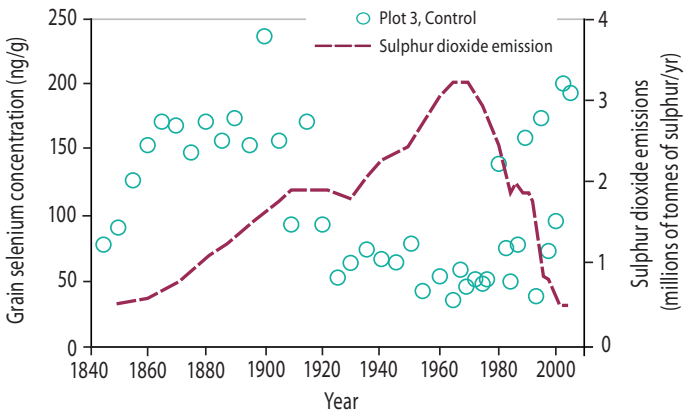

Figure 13.11 Sulphate aerosols (line) and selenium in grain (line) from a plot on the Broadbalk Wheat experiment that has received no fertilisers since 1843. Source: Fan *et al.* (2008). Copyright (2008), reproduced with permission from Elsevier.

13.3.2.2 Pollution

Atmospheric pollution, and specifically the deposition of nitrogen, has been, and continues to be, a major driver of nutrient cycling in terrestrial habitats. Although emissions of reactive nitrogen in the UK have decreased in recent years (RoTAP 2011), there is no evidence of a decline in nitrogen deposition; hence, it continues to impact on nitrogen cycling in UK habitats. In addition, in some terrestrial habitats, significant amounts of nitrogen deposited have accumulated in soil organic matter, thereby reducing both the effects of nitrogen pollution on these habitats and also the export of nitrogen to the environment beyond (Phoenix *et al.* 2004). The trend towards an increase in topsoil carbon:nitrogen ratios suggests that the nitrogen accumulated may be 'diluted' by carbon which may reflect either enhanced nitrogen losses or increased carbon fixation, i.e. plant production (Emmett *et al.* 2010). There is potential for this accumulated nitrogen to be liberated as a result of future land disturbance, changes in land use and/or climate change, but the associated increase in carbon may mitigate against this as nitrogen leaching appears to be closely linked to soil carbon:nitrogen (Emmett *et al.* 2010). As already mentioned, the substantial decline in sulphate deposition, and associated increase in rainfall pH, has contributed to a widespread rise in the pH of soils and waters. This should enhance rates of nutrient cycling in many habitats because many microbial processes that drive nutrient cycling are effective at neutral to slightly acid pH (Bardgett 2005), and because acidification can reduce the diversity of detritivores involved in decomposition processes in water and soil (Gessner *et al.* 2010). Atmospheric sources of nutrients are known to influence freshwater and marine nutrient cycling, but their pattern of addition is complex, being dependent on specific meteorological conditions (Evans *et al.* 2003).

Another driver of soil nutrient cycling is pollution from heavy metals either from sewage sludge, animal manures and other organic and inorganic wastes added to agricultural soils, or from local emissions in urban areas. There is a wealth of evidence suggesting that soil biota is very sensitive to heavy metal stress (Giller *et al.* 1998). For example, Bardgett *et al.* (1994) showed that the abundance of soil microorganisms and nematodes was reduced as a result of heavy metal contamination of pasture, and Giller *et al.* (1989) showed that soil metal pollution dramatically reduces nitrogen-fixation by rhizobia which, in turn, can reduce nitrogen availability in soil. With regards to trends, the recent Countryside Survey found no change in soil concentrations of heavy metals between 1998 and 2007 (Emmett *et al.* 2010).

13.3.2.3 Climate change

There is much uncertainty about the way that climate change will influence natural processes of nutrient cycling and its contribution to past and future trends is unclear. In terrestrial habitats, climate change can impact on nutrient cycling directly by modifying the physical conditions under which biological transformations of nutrients take place,

and indirectly by altering the productivity and composition of plant communities (Bardgett *et al.* 2008; Bardgett & Wardle 2010). For example, warming and changes in moisture conditions can directly impact on the activity of heterotrophic microbes involved in nutrient cycling, as can changes in the frequency of extreme weather events, such as drought, waterlogging and freezing (Schimel *et al.* 2007; Bardgett *et al.* 2008; Whitmore & Whalley 2009). Also, changes in vegetation growth and composition can indirectly influence the abundance and activity of soil organisms, thereby impacting on rates of nutrient cycling and primary production. As an example of this, elevated atmospheric carbon dioxide can increase plant photosynthesis and growth which, in turn, increases the flux of carbon to roots, their symbionts and free-living soil organisms (Bardgett *et al.* 2008). Elevated atmospheric carbon dioxide and changes in weather patterns might also cause shifts in the composition of terrestrial plant communities, with consequences for nutrient cycling (Bardgett *et al.* 2008). Long-term climate change experiments do provide evidence of potential change, with decoupling of carbon and nitrogen cycling (Beier *et al.* 2008), enhanced soil carbon loss (Sowerby *et al.* 2008) and shifts in soil biological activity (Toberman *et al.* 2008) being reported from a long-term climate change experiment in an upland heathland habitat.

Warming has also caused changes in marine fauna generating potential knock-on effects for nutrient and carbon cycling. Warming of water, for example, has caused large changes in the plankton community of deep-sea habitats; many taxa have moved forward their seasonality (i.e. earlier spring blooms) and there has been a progressive shift northward in warmer water zooplankton over the last 50 years (Chapter 12). Also, ocean acidification, which is caused by rising atmospheric carbon dioxide concentrations that drive changes in seawater carbonate chemistry and reduce pH, could affect many marine organisms, with likely consequences for carbon and nutrient cycling. As identified in Chapter 12 there is much uncertainty in this area; however, a recent global meta-analysis suggests that the biological effects of ocean acidification are generally large and negative, but that the variation in sensitivity amongst organisms will have important implications for ecosystem responses (Kroeker *et al.* 2010).

There is much uncertainty about how climate change has, and will continue to, influence nutrient cycling in terrestrial and aquatic habitats due to incomplete knowledge of the mechanisms involved. The impacts of climate change will also vary across UK terrestrial habitats depending on a range of factors, such as vegetation type, soil fertility, soil water conditions and soil type, and in aquatic habitats depending on pre-existing levels of eutrophication. The situation is further complicated in aquatic habitats where long-term nutrient fluctuations are strongly affected by the North Atlantic Oscillation (NAO) and Gulf Stream. For example, long-term (1961 to 1997) inter-annual variations in concentrations of dissolved reactive phosphorus in Blelham Tarn in the Lake District were found to be strongly linked to NAO, mediated through changes in rainfall (George 2002). Likewise, decadal cycles in salinity of the Irish Sea have also been found to be linked to NAO (Evans *et al.* 2003).

13.3.3 Consequences of Change

Nutrient cycling in natural, unpolluted habitats is largely regulated by biological processes; nutrient inputs generally represent only a small fraction of the quantity of nutrients that cycle internally, leading to a relatively closed system maintained by myriad ecological interactions (Bardgett & Wardle 2010). For example, nutrient cycling in natural terrestrial habitats is dependent on an intimate partitioning of nitrogen between the plants and soil microbes over the seasons (Bardgett *et al.* 2005). And, in many habitats, plant nutrient acquisition is strongly dependent on mycorrhizal fungi which associate with the roots of most terrestrial plant species (Smith & Read 2008). Nutrient cycling in rivers and soils is also regulated by detritivore communities which break down dead organic matter (Gessner *et al.* 2010), and by faunal consumers of microbes which regulate nutrient cycling by altering the balance between microbial immobilisation and mineralisation and, therefore, the availability of nutrients for plant uptake (Bardgett & Wardle 2010). As discussed in Chapter 12, nutrient cycling occurs in many components of the marine habitat. In seabed sediments, bacterial processing of nutrients (e.g. nitrification and denitrification) is facilitated by the physical feeding, burrowing and irrigation activity of invertebrates (Covich *et al.* 2004; Olsgard *et al.* 2008), whereas, within the water column, bacterial nutrient cycling is facilitated via food web links with phyto- and zooplankton and fish (Blackford 1997; Proctor *et al.* 2003). Likewise, nutrient cycling takes place between trophic levels and, in the course of bacterial breakdown of detritus (mainly dead algal and plant material), in macroalgal beds and salt marshes. Any disruption of such ecological interactions, for example, through nitrogen enrichment, disturbance or pollution, is likely to have consequences for nutrient cycling in that habitat, although the magnitude of such potential responses is uncertain. Moreover, changes in nutrient cycling that lead to the loss of nutrients to waters and the atmosphere will have consequences for regulating services, as discussed in Chapter 14.

A consequence of enhanced nutrient cycling is a change in the composition and diversity of the terrestrial plant community. Most natural and semi-natural plant communities are nitrogen limited, so increased availability of nitrogen (e.g. through nitrogen deposition or fertiliser addition) will substantially alter their structure and productivity, favouring productive, fast-growing species that are best able to use this added resource at the expense of slower-growing, less competitive species. The implications of this for plant species diversity in UK grasslands are clear from the study of Stevens *et al.* (2004). These authors reported that long-term nitrogen deposition across Britain (ranging from 5–to–35kg N/ha/yr) has significantly reduced species-richness in grassland and that species-richness has declined as a linear function of the rate of nitrogen deposition, with a reduction of 1 species per 4 m^2 for every 2.5kg nitrogen deposited per ha per yr. However, in gradient studies of this kind, there is potential for other factors, such as climate and sulphur deposition, to play role, and it has been argued that the largest changes in plant species composition occurred in the UK in the early part of the 20th Century (Emmett 2007). Indeed, no major changes in plant species composition were detected in a

series of long-term nitrogen addition experiments across the UK, which was attributed to most nitrogen sensitive plant species being already lost prior to the start of the experimental work, i.e. prior to 1990 (Emmett 2007). Nitrogen enrichment can also impact on the diversity and composition of soil biological communities (Bardgett & Wardle 2010), potentially altering rates of decomposition and nutrient cycling, and the ecosystem services that these processes underpin. Indeed, it is well-known that nitrogen enrichment can negatively impact on saprophytic fungi (Donnison *et al.* 2000; Treseder 2008) and the activity of certain extracellular enzymes involved in decomposition processes (Craine *et al.* 2007; Allison *et al.* 2008). It can also change the structure of mycorrhizal fungal communities (Egerton-Warburton & Allen 2000; Frey *et al.* 2004) with unknown consequences for nutrient cycling.

In contrast to conservative or 'closed' systems, provisioning systems exploit nutrient cycling and remove nutrients in products. Agricultural soils are characterised by a high supply of nutrients per unit of organic matter, so it is difficult to farm without loss of nutrients (Goulding 2000). However, enhanced nutrient cycling in provisioning ecosystems could yield additional benefits for regulatory services, especially carbon sequestration. For instance, the application of fertiliser nitrogen could enhance soil carbon storage through increasing plant production and carbon return to soil and by suppressing microbial decomposition of recalcitrant organic matter (Conant *et al.* 2001; Craine *et al.* 2007). Evidence for this is mixed, however, as studies also show that nitrogen fertilisation of agricultural soils can enhance soil organic matter decomposition (Shevtsova *et al.* 2003; Khan *et al.* 2007), and the long-term application of nitrogen to some forests has been shown to have no net effect on soil carbon stocks (Harding & Jokela 2003; Magill *et al.* 2004). There is also a risk that emissions of other greenhouse gases (i.e. nitrous oxide and methane) may increase with nitrogen enrichment, thereby offsetting benefits of enhanced carbon dioxide uptake (Liu & Greaver 2009). In view of this uncertainty, it is not yet possible to make sweeping statements about how soil carbon sinks will respond to nitrogen enrichment (Reay *et al.* 2008).

The consequences of ocean acidification for nutrient cycling are uncertain (Chapter 12). But the consequence of the reduction of acidity and the decline in phosphorus content of freshwaters will be an improvement in water quality due to reduced eutrophication. This will not only lead to reduced treatment costs for downstream users, but it will also increase amenity value of waters due to reductions in toxic algal blooms and physical blocking of waterways. In terrestrial habitats such as grasslands, high soil phosphorus concentrations can constrain the restoration of plant diversity (Smith *et al.* 2008b); hence, widespread reductions in soil phosphorus could, in some habitats, yield added benefits for biodiversity conservation.

13.3.4 Options for Sustainable Management

A key goal for sustainable management of semi-natural and natural habitats is the protection of ecological interactions that contribute to 'closed' systems of nutrient cycling, characterised by efficient use of nutrients by plants and microbes, and low levels of nutrient loss to waters and the atmosphere. Following on from this, a key goal for sustainable management of provisioning systems, such as agricultural land, is to develop management strategies that enhance reliance on natural processes of nutrient cycling and plant nutrient acquisition, and which minimise nutrient loss to waters and the atmosphere. Several management options could help achieve this goal including more efficient use of fertilisers, greater use of perennial crops, legumes and crop mixtures, and the use of intercropping, which has been found to reduce losses of nitrogen from systems (Whitmore & Schröder 1997). In addition, the exploitation of deep-rooting plants capable of retaining or capturing nutrients, or those which can obtain phosphorus via acid exudation or by means of mycorrhizal fungi, offers potential in terms of the sustainable management of nutrients. Management strategies that enhance the abundance of fungi relative to bacteria in soil, such as no-till agriculture and reduced fertiliser use, also offer the potential to improve the efficiency of nutrient cycling, including the retention of nitrogen and phosphorus in soil (de Vries *et al.* 2006; Gordon *et al.* 2008). In general, our understanding of the mechanisms by which plant-soil interactions regulate nutrient cycling is limited. Hence, new research is needed to better exploit the potential for different plants to access nutrients efficiently from soil via their interactions with soil organisms, and to develop management strategies that increase reliance on soil biological processes for the provision of available plant nutrients for crop growth.

Finally, the contribution of soil erosion to nutrient cycling and the transfer of nutrients to lakes, rivers and the ocean are not widely appreciated. Soil erosion contributes greatly to nutrient loss from land, especially phosphorus, and to the transport of nutrients to the ocean (Quinton *et al.* 2010). It was recently estimated that the amount of nitrogen moved by erosion globally is in the order of 37–75 Tg N/yr, which is of the same order of magnitude as the 98 Tg N/yr nitrogen applied to agricultural land as chemical fertilisers (Quinton *et al.* 2010). Also, soil erosion-driven fluxes of phosphorus have been estimated to be similar in magnitude to the amount of fertiliser phosphorus added to agricultural land per year (Quinton *et al.* 2010). Management strategies aimed at preventing soil erosion will therefore have major implications for nutrient cycling and the regulating and provisioning services that it underpins.

13.3.5 Knowledge Gaps

Although there have been significant advances in our mechanistic understanding of nutrient cycling and the response of nutrient cycles to various environmental and management drivers, significant gaps remain. One of the biggest challenges concerns the need to better understand the biological mechanisms that regulate nutrient cycling in different habitats and the role that biodiversity plays in nutrient cycling in soil and water. In particular, the contribution of soil microbes to the functioning of ecosystems is still poorly understood, largely because more than 95% of microbes are unable to be cultured on conventional media (Van der Heijden *et al.* 2008; Singh *et al.* 2010). And so, our understanding of the functional consequences of changes in the diversity and

composition of complex soil communities remains poor (Bardgett & Wardle 2010). Such an understanding is essential for developing future land management strategies aimed at enhancing reliance on natural processes of nutrient cycling and plant nutrient acquisition, and for minimising nutrient loss from land to waters and the atmosphere.

Another major gap in understanding concerns the impact of climate change and other drivers (e.g. nitrogen deposition) on nutrient cycles. In particular, much remains to be learned about the response of nutrient cycles to climate change which involves many direct and indirect mechanisms operating differently across habitats. A particular gap in understanding concerns the effects of climate change-driven alterations in the allocation of carbon to roots, and the transfer of plant carbon to soil, which can stimulate the abundance and activity of soil microbes, enhancing the mineralisation of both recent and old soil organic carbon (i.e. priming), and therefore increase carbon loss from soil (Bardgett *et al.* 2008). Indeed, several free-air carbon dioxide enrichment (FACE) experiments have shown that elevated carbon dioxide can lead to substantial increases in root biomass and soil respiration (Pritchard *et al.* 2008; Jackson *et al.* 2009), and that, in general, below-ground responses to elevated carbon dioxide are often greater than above-ground responses in the same systems (Jackson *et al.* 2009). Mycorrhizal fungi also act as a significant sink for recent plant photosynthate (Högberg & Read 2006), and increased supply of photosynthate carbon to these fungi under elevated carbon dioxide can stimulate their growth, especially under conditions of low nutrient availability (Klironomos *et al.* 1997; Staddon *et al.* 2004), with unknown consequences for nutrient cycling. It is important to note that drivers other than climate change can strongly affect organisms involved in nutrient cycles. As a consequence, it is becoming increasingly apparent that our ability to predict future responses of nutrient cycles to global change requires a greater understanding of the simultaneous effects of multiple drivers (Tylianakis *et al.* 2008; Bardgett *et al.* 2008). Thus, understanding how multiple drivers acting simultaneously affect nutrient cycles represents a major research challenge for the future.

Many gaps in understanding exist concerning the fate and transfer of anthropogenic nitrogen in terrestrial and aquatic habitats, and its impact on nutrient cycles and other ecosystem services. For instance, little is known about the mechanisms involved in the retention of anthropogenic nitrogen in soils, although it is evident that microbes can act as a major sink for nitrogen and that they can regulate the transfer of nitrogen to soil organic matter, plants and aquatic habitats. Also, nitrogen enrichment can impact on other ecosystem services, especially carbon sequestration, although much remains unknown about the mechanisms involved and how the effects of nitrogen on carbon cycling vary across habitats. Our understanding of phosphorus cycling is also limited, especially with regards to how plants and microbes interact to access soil phosphorus, and how both nitrogen and phosphorus cycling are coupled. In general, there have been relatively few studies on the interactions between nutrients or between environmental stresses and nutrient cycling. Thus, the mechanisms involved in the widespread decline in phosphorus availability across UK habitats, and the implications for other ecosystem services, are poorly understood.

13.4 The Water Cycle

13.4.1 Condition, Status and Trends

In terms of supporting ecosystem services, the water cycle is most appropriately considered with respect to the major water fluxes (rainfall, evapotranspiration, river flow) and the major water storages (soil, groundwater, lakes) that combine to determine the availability of water in time and space. Fluxes and pathways that move water between the major storages are also important, as is the issue of water quality—poor chemical and/or microbiological quality can render the water effectively unavailable for supporting some services. Hydro-meteorological monitoring in the UK allows assessments of both rainfall and runoff (**Table 13.2**) to be made with reasonable accuracy (Marsh & Sanderson 2006).

Spatial variation in average rainfall across the UK is exceptionally high: annual totals can exceed 5,000mm in parts of the western Highlands of Scotland, whilst totals in the driest parts of lowland England can be an order of magnitude less (**Figure 13.12**). National rainfall assessments for England and Wales are available back to 1766 (**Figure 13.13a**) and, although rainfall for the last 30 years modestly exceeds the average for the preceding record, there is no overall trend (Alexander & Jones 2001). For Scotland, precipitation has increased substantially since a notably dry episode in the early 1970s; over the last 30 years, it is around 5% greater than the preceding average (**Figure 13.13b**) as a result of increased winter precipitation, particularly in the western Highlands. A change in rainfall seasonality has also been observed, with wet winters (November to April) more common during the last 30 years than during much of the 19th Century when summer rainfall totals (May to October) were generally higher.

Estimation of the flux of water lost in gaseous form as evapotranspiration (ET) largely represents rainfall minus runoff (**Table 13.2**). The flux is highly variable over the scale of a few metres and depends on factors such as plant cover and surface wetness. The long-term catchment experiment at Plynlimon is the best record of rainfall and runoff available

Table 13.2 UK water balance (mm), 1971–2000.
Source: rainfall data based on data from the Met Office National Climate Information Centre.

	Rainfall	Runoff	Rainfall-Runoff
GB	1,086	650	436
England and Wales	896	455	441
Scotland	1,440	1,100	340
Northern Ireland	1,111	679	432

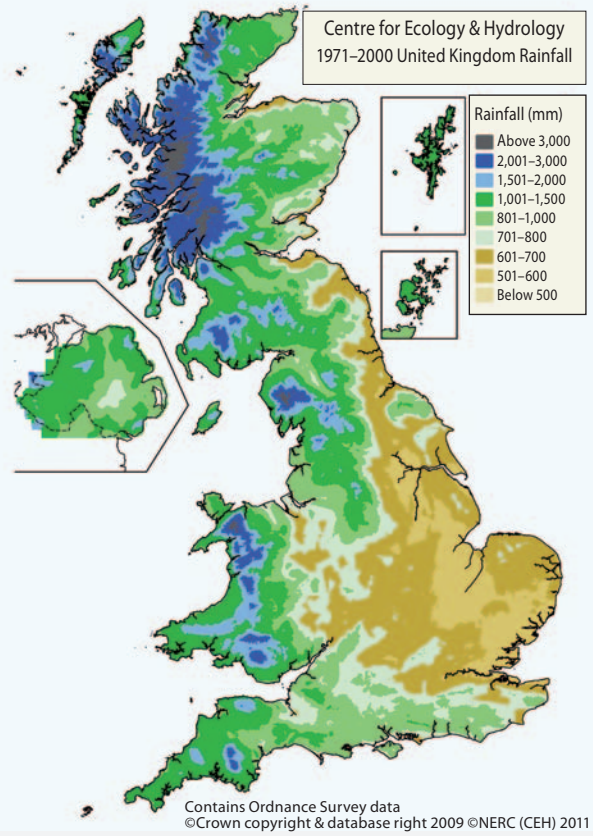

Figure 13.12 UK average annual total precipitation, 1971 to 2000. Source: data from Met Office National Climate Information Centre.

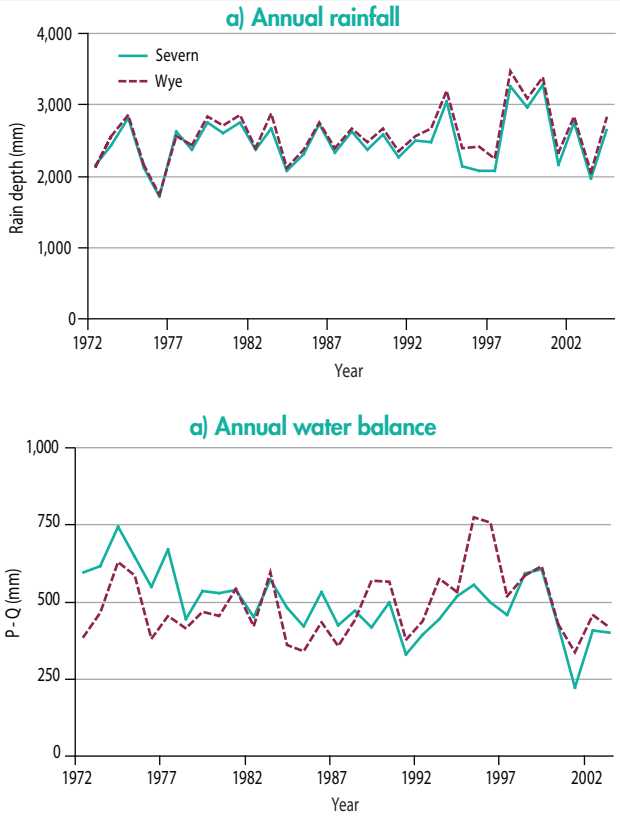

Figure 13.14 a) Long-term rainfall; and b) water balance (evapotranspiration) from the forested (Severn) and moorland (Wye) catchments at Plynlimon. Source: after Robinson *et al.* (submitted).

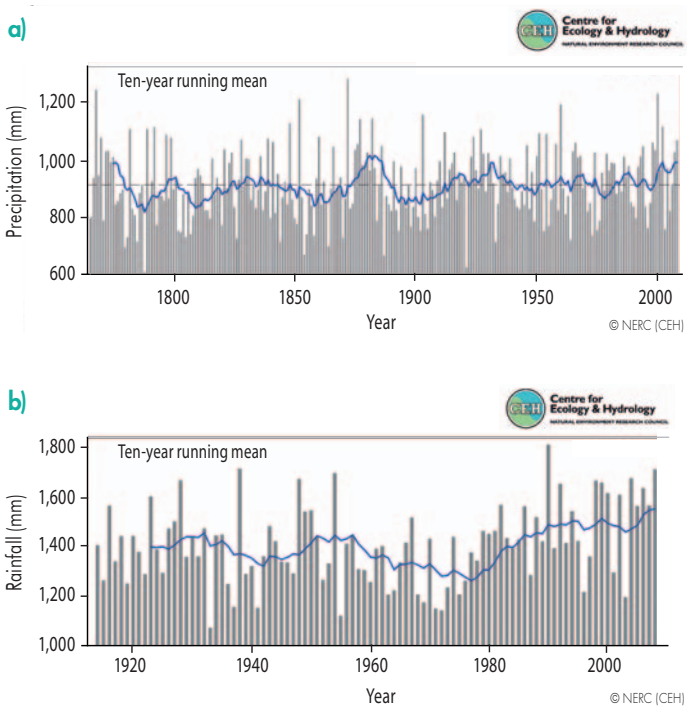

Figure 13.13 Annual total precipitation in a) England and Wales, 1766 to 2008; and b) Scotland annual total precipitation, 1914 to 2008. Source: data from a) Met Office Hadley Centre / UEA Climate Research Unit; and b) Met Office National Climate Information Centre.

(**Figure 13.14**) and shows the impact of a maturing pine forest plantation promoting increased evaporative losses relative to a nearby moorland (Marc & Robinson 2007). The UK's Met Office Rainfall and Evaporation Calculation System (MORECS) provides assessments of potential (**Figure 13.15a**) and actual (**Figure 13.15b**) evaporative losses for 40 km squares throughout Great Britain. These assessments indicate that, on average, over 40% of UK rainfall is lost to evaporation, although the proportion varies greatly regionally, reaching around 80% in the driest parts of the English Lowlands (Hough & Jones 1997).

River flows integrate the precipitation and evaporation interactions at the basin scale and, therefore, measured runoff represents the most appropriate variable upon which to assess overall water resources and hydrological variability in space and time. Many UK river flow regimes, however, are profoundly affected by artificial influences such as abstractions and river regulation. Given the limited length of most river flow records and the climatic volatility of the last 30 to 40 years, the presence or absence of hydrological trends can be very sensitive to the length of record under review. There exists little indication of trend in total annual runoff from England and Wales between 1961 and 2008, although the recent past has been characterised by notable year-on-year variability (Hannaford & Marsh 2006). For Scotland, runoff has increased markedly following the early 1970s, but with limited change over the last 25 years (Hannaford & Marsh 2006).

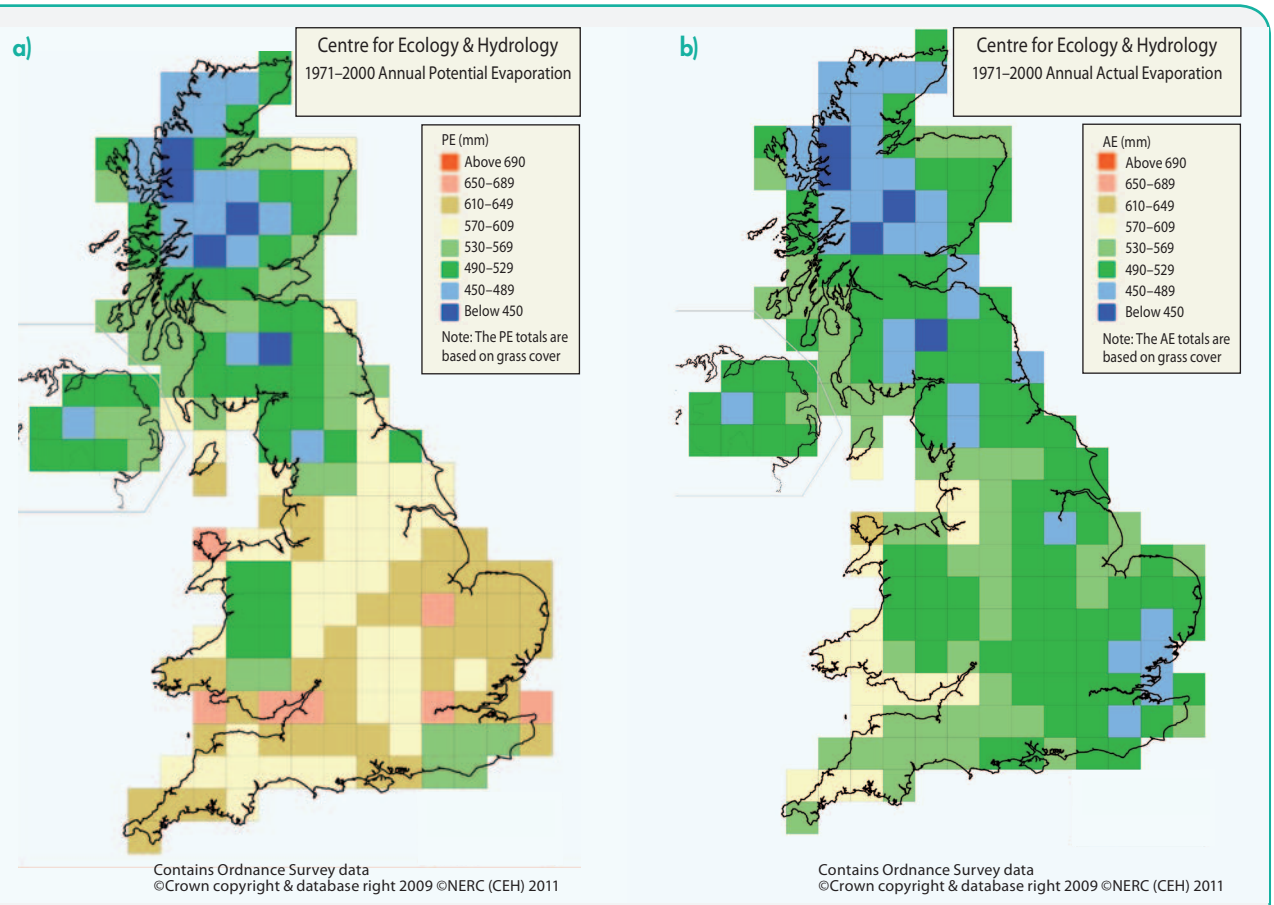

Figure 13.15 Great Britain average a) annual Potential Evaporation (PE); and b) Actual Evaporation (AE) totals, 1971 to 2000. Source: Met Office Rainfall and Evapotranspiration Calculation System.

Regional variations in runoff patterns across the UK are notably large, reflecting the diversity of the UK in terms of its climate, geology, land use and water utilisation patterns. The network of UK Benchmark Catchments (Hannaford & Marsh 2006) allows spatial responses to be assessed from the early 1970s, although particular caution needs to be exercised when interpreting trends over the last 30 to 40 years since the 1970s were a quiescent period in hydrological terms, while the period 1998 to 2002 is the wettest five-year sequence on record. Trend analysis of annual runoff over the period 1973 to 2002 at the UK Benchmark Catchments indicates significant increases in runoff (Hannaford & Marsh 2008) for a number of northern and western rivers (**Figure 13.16**). Corresponding analyses for the limited number of river flow time series over 70 years in length suggest that such trends are considerably less compelling as the time-span examined increases. Nonetheless, a modest tendency towards recent increase in runoff can be recognised. Low flows (30-day minima) at the Benchmark Catchments over the period 1973 to 2002 indicate little evidence of a decrease in low flows across the UK (Hannaford & Marsh 2006).

Groundwater storage (i.e. water located beneath the ground surface) provides a significant component of water resources used in the UK. Aquifers are generally replenished by winter rainfall when soil moisture deficits are negligible, although intense rainfall events can lead to localised recharge at any time of the year. Groundwater levels within aquifers represent a dynamic balance between

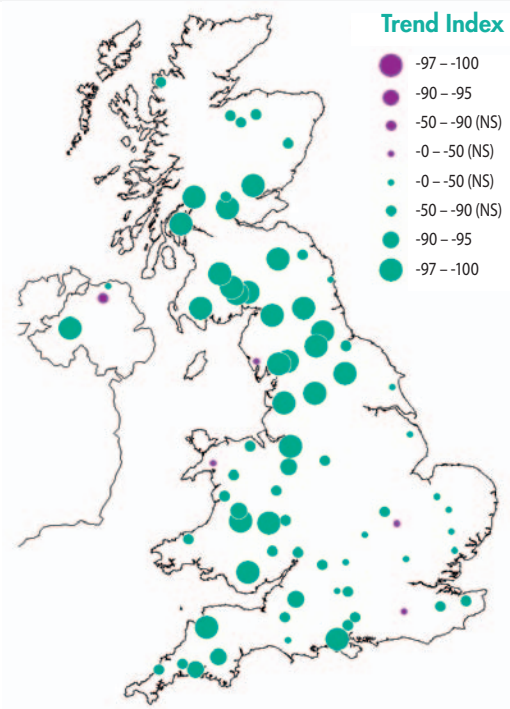

Figure 13.16 Trends in 10-day maximum flows for UK Benchmark Catchments, 1973 to 2002. Purple circles denote negative trends, turquoise circles positive trends, with graduated symbols representing statistical significance. Source: Hannaford & Marsh (2008). Copyright © 2007 Royal Meteorological Society. Reproduced with permission of Blackwell Publishing Ltd.

natural recharge, discharge to rivers and the sea, and anthropogenic abstractions, which often represent a large proportion of the water balance. The longest continuous record of groundwater level in the UK dates back to 1838 at Chilgrove in West Sussex. A drought index, representing cumulative departures from mean monthly levels, shows no consistent trends over this period, but distinct patterns of activity or quiescence at decadal scales exists. Trends in groundwater levels over decadal timescales most often result from changes in historical abstraction, for instance commencement or cessation of pumping for public supply, industry or mine dewatering.

13.4.2 Drivers of Change

The water cycle is extremely dynamic and sensitive to a range of drivers: impacts on any one major water store or flux can lead to significant changes propagating throughout the system. The major driver of long-term changes in the quantity of water in rivers, soils and groundwater is human activity such as changes in land use (including sealing due to urbanisation), drainage of agricultural land, development of impoundments and supply structures, and structural changes to rivers and water abstraction. In addition, acute or short-term changes are largely driven by cyclical changes in weather over a period of several years (e.g. The North Atlantic Oscillation). Extreme weather events, such as droughts (lasting one or two years) and floods (lasting days or weeks), can also have long-lived impacts on the water environment and water resource availability.

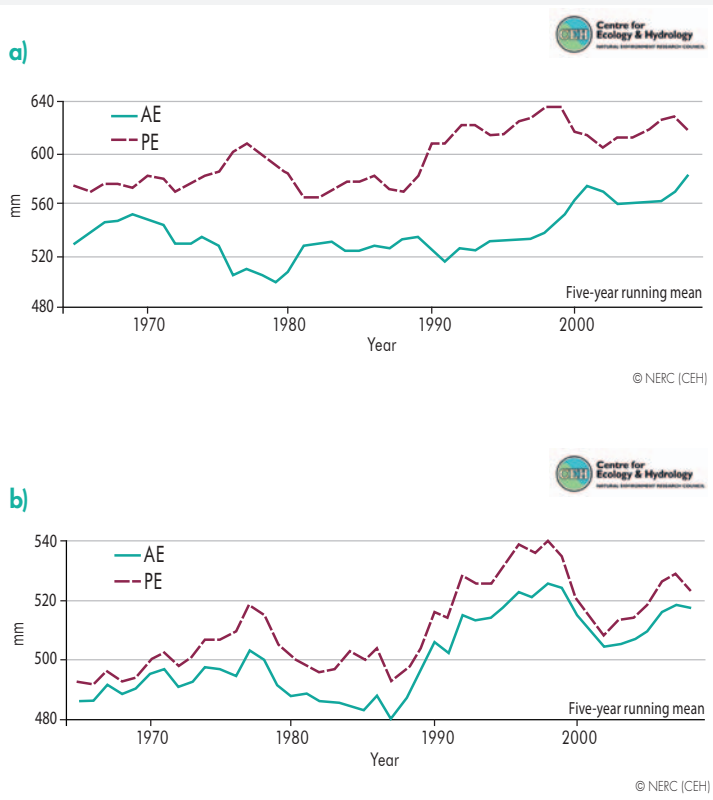

Figure 13.17 Annual Potential Evaporation (PE) and Actual Evaporation (AE) totals in a) England and Wales; and b) Scotland. Source: Met Office Rainfall and Evapotranspiration Calculation System.

13.4.2.1 Land use and climate

The importance of future drivers will differ between habitats. For example, the water cycle in mountain habitats is most impacted by changes in temperature and rainfall which influence rates of evapotranspiration and, therefore, soil moisture and river flow regime. It is also impacted by changes in atmospheric deposition, especially of nitrogen. In semi-natural grassland areas, river flows and soil moisture regimes reflect changes in precipitation and temperature, but significant land use change can engulf this signal. In farmlands, pressures from increased abstraction for irrigation and agrochemical runoff are likely to have the greatest impact on the water cycle. The impacts of increasing populations in currently water stressed areas, such as the south-east of England, are further likely to promote increased competition for the available water resource between human and environmental needs.

13.4.2.2 Climate change

All of the above drivers are likely to be amplified in the future as the UK adapts to an increasing population, changing energy mix (e.g. increased importance of biofuels, hydro-power) and future climate change. Climate change alone is predicted to influence patterns of water movement both spatially and temporally, including greater frequency of extreme events and long-term changes in precipitation and evaporation. Climate change may also amplify effects of other drivers such as agriculture, with potentially more water resources required for irrigation and the replacement of conventional crops with dedicated energy crops. Such crops will result in increased evaporation rates, but reduced risk to water quality due to low inputs of agrochemicals. As an example, annual average evaporation loss for England and Wales has been observed to have gradually increased from around 500 to 550 mm over the period 1980 to 2005. This reflects an increase in average temperatures across the UK of around 1.0°C over the last 30 to 40 years (**Figure 13.17a,b**).

13.4.3 Consequences of Change

The consequences of future climate change in the UK are predicted to have a profound effect on the timing and magnitude of rainfall and runoff patterns. On average, the UK is likely to experience warmer, drier summers, and warmer, wetter winters (UKCP09). This will cause changes in annual river flow, although changes will be catchment-specific and will depend upon topography, soils, land use and geology. In addition, it is predicted that seasonal changes in the duration and intensity of rainfall will promote an increased occurrence of extreme floods, yet there is no clear understanding about change in size, or even direction, of flood-flow magnitudes in the UK. Changes in flood flows will also be catchment-specific, being driven by hydrological variables, such as geology, and the seasonal distribution of rainfall (Reynard *et al.* 2007). With respect to low-flows and drought periods, the projected increase in mean precipitation in winter and decrease in summer

months may lead to an increase in frequency of short-term summer drought in most water resource regions, except Scotland and Northern Ireland. However, the uncertainty associated with such changes is immense (Blenkinsop & Fowler 2007).

Assessing the consequences of climate change on the availability and flux of freshwater in response to future climate change, however, is more complex than simply quantifying changes in rainfall, evapotranspiration and runoff. In most UK river catchments, human activities have historically driven significant changes to the water cycle through increased abstraction for water supply, agriculture, industry, and to sustain an increasing human population, and through changes in drainage in urban areas and for navigation management, flood defence and re-alignment (Ferrier *et al.* 2010). The water cycle is conventionally managed with the catchment as a central concept in water resource planning. The available water resources must support all demand within the catchment, and often outside the catchment area as well, including potable supply, irrigation, cooling and for ecosystem services. Activities and decisions made within catchment systems and relating to any one resource demand, will ultimately impact on the nature and function of downstream waterbodies, be they groundwaters, lakes, rivers, wetlands, estuaries or coastal zones. Traditionally, water resource management systems have been operated under the assumption of stationarity, where the statistical distribution (i.e. mean, standard deviation, etc.) of observations do not change over time. Hydrologists now face a future where assumptions about key parameters can no longer be based on historical records (Milly *et al.* 2008), making the consequences of change more difficult to predict and to manage.

13.4.4 Options for Sustainable Management

The sustainable management of water relies fundamentally upon the assessment of equitable allocation of available resources to all sectors requiring them. Protecting and providing for the water requirement for potable supply, industry and agriculture have conventionally formed the basis of management legislation, whilst water for ecosystem services has not been formally quantified. As a result, the ecosystem gets what resource remains. Of course, this can lead to problems for the provision of ecosystem services in areas of high anthropogenic water demand, particularly at times of water shortage, usually during the summer or prolonged drought periods.

The Water Framework Directive (WFD; EU 2000) provides a legislative platform upon which to base a more holistic approach to the water cycle, including the maintenance of ecological quality. Its objectives are: to protect and, where necessary, to improve the quality of all inland and coastal waters, groundwater and associated wetlands; to promote the sustainable use of water; to enhance protection and improvement of the aquatic environment; and to lessen the effects of flooding and drought. A key concept underlying the WFD is that of integration of environmental objectives, combining quality, ecological and quantity objectives, and all water uses, functions and values, into a common policy framework. This basically translates as water for the environment, water for health and human consumption, water for economic sectors, transport and leisure, and water as a social good. Conflicts in management of water resources, however, are common and are likely to become more commonplace in a future where rainfall totals are decreased, demand for water increases and water resources are scarcer.

Managing flooding presents further conflicts for holistic water management. For example, it is widely recognised that raising water levels in floodplain wetlands can have generally positive outcomes for ecosystem services such as carbon sequestration. However, it may also have detrimental effects, for instance, on flood storage. It has been calculated, for example, that for the North Drain catchment in the Somerset Levels and Moors, raising ditch and drain water levels to field level would remove 3.6 million m^3 of floodwater storage. This is equivalent to about 84% of the median annual maximum flood for the North Drain catchment (Acreman *et al.* 2007).

13.4.5 Knowledge Gaps

In general, our understanding of the water cycle is well advanced, and this has provided the basis for the establishment of predictive models capable of describing the spatial and temporal variation in the water cycle, and for predicting future changes in the water cycle in response to key drivers. When tested against observations of river discharge over time, however, hydrological models are often incapable of accurately simulating both the extreme storm-flow and base-flow of a catchment. This implies the need to elaborate the flow paths between groundwater, soil water and river channels within catchments. Such an understanding requires very detailed measurements of water stores and fluxes in space and time. The necessity for such a development lies in the link between ecosystem function and water; this is unlikely to relate to either simple high or low flow extremes, or to simple mean flows over a time period, but rather to periods of time for which critical flows are exceeded. These critical flows may also vary for different seasons of the year.

Soil water remains perhaps the most elusive store to quantify because measurements are sparse and tend to relate only to individual points within a catchment. Again, models have filled this gap, but without appropriate data for testing them, how applicable they are remains uncertain. Therefore, future studies need to better understand the dynamics of soil water and the influence of various drivers, such as climate change and land use, on soil water fluxes. Such an understanding also has implications for nutrient cycling, since most biological processes that govern nutrient and carbon cycling are strongly affected by soil moisture.

Another challenge concerns the need to reduce uncertainty, and increase resolution of, hydrological models and their outputs. This is crucial because our ability to predict future variability in the water cycle in space and time relies on the hydrological models being coupled to climate models capable of producing rainfall fields at an appropriate scale and time resolution. This is an area where further research is required.

13.5 Primary Production

13.5.1 Condition, Status and Trends

Primary production is the fixation by photosynthesis of either atmospheric or aquatic carbon dioxide, or its assimilation as organic compounds by plants and algae. Only a proportion of carbon dioxide fixed by photosynthetic organisms is retained; the rest is lost through the respiration processes that are required to maintain existing biomass. The overall amount of organic carbon fixed is described as gross primary production (GPP), whilst the amount retained after respiration is defined as net primary production (NPP). It is NPP that is of most relevance for ecosystem service delivery: it is available for food and timber harvesting in managed systems, provides the foundation of food webs in semi-natural and natural ecosystems, and underpins climate regulation by removing carbon dioxide from the atmosphere.

Major controls on primary production include light, temperature and water, the availability of nutrients (especially nitrogen and phosphorus), community diversity and composition, and the intensity of management to which vegetation is subjected. Response functions vary for each of these controls. For example, too little or too much water results in reduced rates of plant primary production in terrestrial systems, whilst there is typically a humpback relationship between plant productivity and diversity, and between disturbance and diversity (Grime 1979; Al Mufti *et al.* 1979). The relative importance of different factors that control primary production will also differ across habitats. For example, land management is the primary control on primary production in agricultural habitats, whilst water table height might be the primary control of primary production in wetland habitats. In marine habitats, most primary production is carried out by microbial communities in the phytoplankton, which include diatoms, coccolithophores and cyanobacteria.

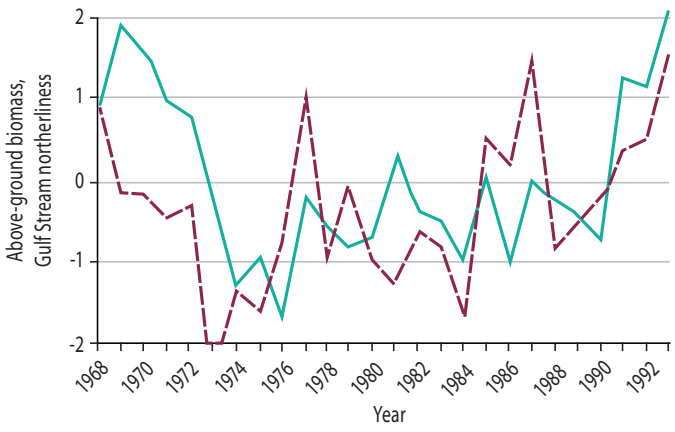

Figure 13.18 Standardised mean annual variations in above-ground biomass in road-verges in Bibury, Gloucestershire (solid line) and Gulf Stream northerliness index in the previous spring–summer months (dashed line). Source: Willis *et al.* (1995). Copyright © 2005 Nordic Society Oikos. Reproduced with permission of Blackwell Publishing Ltd.

Quantifying the status and trends of primary production is problematic because it is not possible to measure it directly, other than at local scales. As a consequence, data on the status and trends of primary production in the UK is limited. Particular problems include our inability to quantify production. Hence, knowledge is lacking about the amount of fixed carbon that is allocated below-ground to roots, which varies greatly across habitats (Jackson *et al.* 1996). This plays a significant role in soil carbon sequestration which is one of the main services derived from primary production (De Deyn *et al.* 2008). There is also a lack of information about the fate of organic and inorganic carbon fluxes to surface waters, which, if not included, can lead to major underestimations of primary production in aquatic systems (Evans *et al.* 2005). Finally, incomplete knowledge of response functions to different climate, soil, ecological and management variables, both singly and in combination and across all habitats, limits our ability to upscale primary production estimates in both time and space. These problems have led to a range of approaches and surrogate measures being employed, either singly or in combination, to provide estimates of primary production at a range of scales from plot to catchment, regional to national levels. These approaches include: net ecosystem exchange of carbon dioxide (NEE), which is the balance between net uptake of carbon dioxide by photosynthesis and its loss by respiration; net ecosystem productivity (NEP); plant standing biomass or biomass increments; crop, herbage and timber yields (Jenkinson *et al.* 1994); carbon accumulation rates; chlorophyll *a* concentrations and remotely sensed absorption by plants and algae. These approaches often include many assumptions and yield estimates often with limited justification (Lovett *et al.* 2006).

The earliest attempts to estimate primary production were made by geographers based on regressions of temperature data computed to a simple annual measure of actual evapotranspiration (AET), which was then regressed to field measurements of standing plant biomass (Running *et al.* 2004). This type of measurement can provide clear links to climate variables such as that demonstrated in biomass changes in a 20-year record of road verge vegetation by Willis *et al.* (1995) (**Figure 13.18**). These authors found that total vegetation productivity of above-ground vegetation was positively correlated with minimum spring temperature and that, in general, those plant species favoured by environmental stress or disturbance were promoted following warm dry springs and summers, whereas those preferring more productive conditions were promoted following a wet growing season (Willis *et al.* 1995). In the UK's longest record of herbage yields, the Park Grass Continuous Hay experiment at Rothamsted, climate variables explained between 12–21% of variability in grass yield in both unfertilised and fertilised plots over a 100-year time period (Jenkinson *et al.* 1994).

In forests, annual stem increment data, plus litter fall and assumptions for below-ground root production, are often used in combination with remotely sensed plant biomass data to derive net primary production values. Forest management aims at maximising timber yields and trees are harvested when rates of growth start to decline due to maturation; this typically occurs between 40 to 50 years for

monoculture softwoods such as spruce. Native woodlands are often less intensively managed to enhance other services such as biodiversity and recreation. Current forestry production rates in the UK have recently been estimated to be responsible for removing 15 Mt of carbon dioxide from the atmosphere every year with a total stock of carbon of 790 Mt of carbon in trees and forest soils (Read *et al.* 2009; Chapter 8 and Chapter 15).

The use of eddy-covariance in terrestrial habitats to measure the total exchange of carbon dioxide (i.e. NEE) has been employed to estimate landscape-scale spatial and temporal patterns of primary production. For example, work on ecosystem carbon dioxide exchange in forests across Europe identified the clear negative impact of the summer drought in 2003 on NPP. This trend was also observed for other habitats using remotely sensed radiation measures and country-level crop yields (Ciais *et al.* 2005). However, these measurements omit DOC, POC and inorganic carbon fluxes to aquatic systems (as highlighted above) and, therefore, underestimate primary production.

Earth observation provides another tool to estimate primary production based on the amount of photosynthetically active radiation available, the fraction of photosynthetically active radiation absorbed, estimated using the Normalised Difference Vegetation Index (NDVI), the amount of leaf area present to absorb solar energy, and a conversion efficiency factor (Running *et al.* 2004). The use of NDVI at the UK scale has identified the complex spatial pattern of leaf area that is required for primary production and how it is driven by a range of inherent landscape properties including soil type, topography and altitude, as well as land management practices. In marine systems, the patterns of primary production of coastal waters can be determined as a product from Earth observation methods and the spatial and temporal patterns of primary production can be identified in UK marine waters (**Figure 13.19**). The images indicate how the spring blooms of primary productivity start on the shelf and then move into deeper waters in the ocean. The Plymouth Marine Laboratory (PML) produces estimates of net primary production for the seas around the UK (daily estimates) and globally (monthly estimates). These use inputs of satellite-derived chlorophyll *a* and sea-surface temperature, and measured or modelled irradiance. The estimates of primary production use the model of Smyth *et al.* (2005) which is forced by phytoplankton chlorophyll *a* which absorbs light for photosynthesis, temperature which affects the rate of growth of the phytoplankton, and irradiance on the sea-surface and its attenuation with depth, which depends on the optical constituents in the water. The current model allows for in-water absorption by water, phytoplankton and its associated by-products, including co-varying coloured dissolved organic matter (CDOM). However, the model does not account for the effects of suspended particulate matter (notably in the Thames Estuary, southern North Sea, Bristol Channel) nor CDOM from riverine sources (such as in the Baltic outflow along the Norwegian coast or in Liverpool Bay). In these areas, primary production is likely to be overestimated. Work at the Plymouth Marine Laboratory is aiming to improve these coastal estimates.

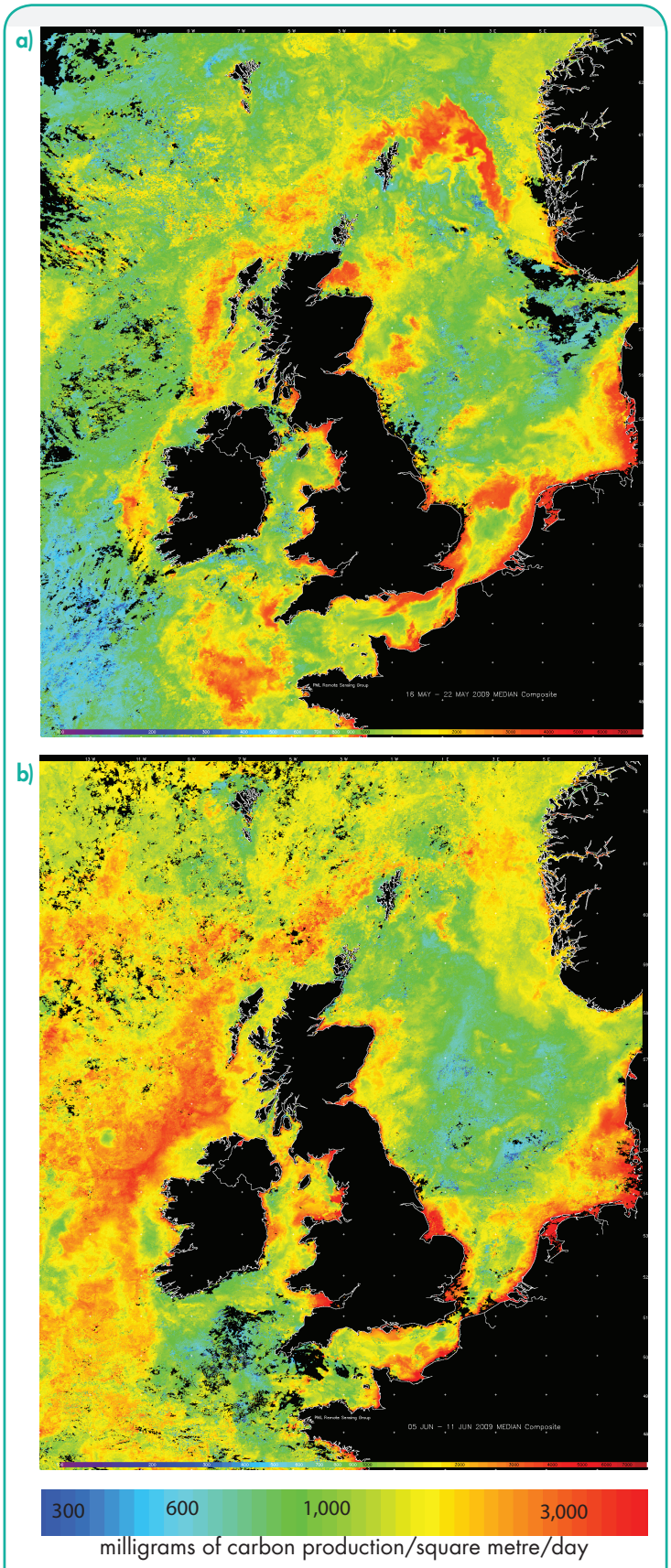

Figure 13.19 Seven day 'composites' produced from the NASA MODIS Aqua instrument received at NEODAAS-Dundee and processed at NEODAAS-Plymouth for a) 16–22 May; and b) 5–11 June 2009. The estimates of primary production use the model of Smyth *et al.* (2005) and are probably overestimated close to the coast in the southern North Sea.

In freshwaters, primary production is estimated from the surrogate phytoplankton chlorophyll *a* concentration. Primary productivity in most freshwater systems was thought to be primarily phosphorus limited but, as discussed previously, there is now strong evidence that nitrogen limitation, and nitrogen and phosphorus co-limitation, is widespread, especially in UK upland environments (Maberly *et al.* 2002). Long-term historical trends of 36 years are available for four lakes in the Lake District; here there is some evidence of suppression of chlorophyll *a* concentration with higher rainfall in small lakes with shorter residence times, but there was no overall trend in chlorophyll *a* concentrations (George *et al.* 2004).

As can be seen above, modelling provides a valuable tool to upscale and integrate measurements. It can also be used to test our current understanding of controls of primary production in the UK and forecast changes under different climate, pollution and management scenarios. For example, the JULES model (The Joint UK Land Environment Simulator) can produce a range of outputs, including primary production. However, a great deal of testing and development of the JULES model is required before its outputs can be used with any confidence. Models are also used in the marine environment, for example, ERSEM (European Regional Seas Ecosystem Model) coupled with the hydrographic model POLCOMS (The Proudman Oceanographic Laboratory Coastal Ocean Modelling System), which is now being used by the Met Office to provide marine system forecasting (www.ncof.co.uk/Coastal-Seas-Modelling.html).

13.5.2 Drivers of Change

In addition to intended modifications of primary production for food and fibre production, human activities have also had unintended effects on primary production through: (i) climate change; (ii) change in light levels due to reduced sulphur emissions and aerosol formation; (iii) various aspects of air quality including continued elevated reactive nitrogen in the atmosphere, reduced acidity and increased ozone; and (iv) nutrient release into aquatic systems (**Figure 13.20**).

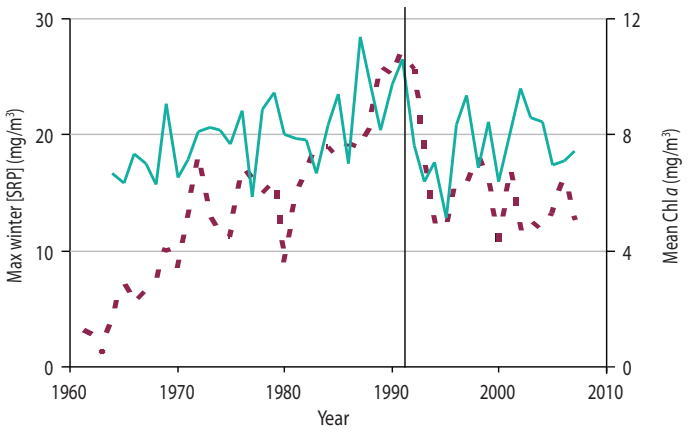

Figure 13.20 Long-term changes in the maximum winter concentration of soluble reactive phosphorus (SRP, dotted line) and annual mean concentration of phytoplankton chlorophyll *a* (solid line) in the South Basin of Windermere, Cumbria. The vertical line show the start of tertiary P-removal in 1992 at the two wastewater treatment works that discharge directly into Windermere. Source: data © NERC (CEH), © FBA (Freshwater Biological Association).

13.5.2.1 Land use

The main driver of change of terrestrial primary production has been, and continues to be, the production of specific food and fibre products and an associated increase in overall productivity through land use and management. This has been highly successful, resulting in an exponential rise in food production over the last 150 years, particularly through the development and widespread use of nitrogenous fertilisers in combination with new crop varieties (Goulding *et al.* 2008; **Figure 13.21**). Other successful management practices include a wide range of activities in different habitats, such as improved grazing management, genetics and crop improvements, fertiliser use, liming and drainage. For much of the second half of the 20th Century, subsidies often dominated over economic market forces in ensuring continued focus on food production, although more recent policies directed at biodiversity conservation have led to extensification of management and reduced productivity in some habitats, such as agriculturally improved grassland. As discussed in Section 13.1, changes in land use, for instance increased grazing and fertiliser use, can also instigate plant-soil feedbacks that promote nutrient cycling and, therefore, plant production by altering the composition of the soil microbial community (Bardgett & McAlister 1999; Smith *et al.* 2008b).

13.5.2.2 Nitrogen deposition

Atmospheric nitrogen deposition is thought to be a major driver of change in primary productivity in UK semi-natural habitats. But evidence of its importance is contradictory, with no trends in primary productivity being observed for the longest agricultural herbage record at Rothamsted (Jenkinson *et al.* 1994), but increased primary productivity being measured or modelled in many forest systems (de Vries *et al.* 2009). Some soils appear to have a very high capacity to accumulate and retain nitrogen in stable organic matter pools, thereby reducing nitrogen export and the harmful effects of pollution. Indeed, most mineral nitrogen entering soils, whether from deposition or fertilisers, is rapidly immobilised by soil microbes and, in strongly nitrogen limited habitats, is subsequently transferred to plants and/or stable, non-microbial organic matter pools (Zogg *et al.* 2000; Bardgett *et al.* 2003). Indeed, the primary control on nitrogen retention and leaching to freshwaters appears to be the potential for vegetation uptake (e.g. high in an aggrading forest) and relative and absolute amounts of carbon and nitrogen in soil, which determines its capacity to sequester more nitrogen (Emmett 2007). Despite the above, long-term changes in vegetation diversity and composition across UK habitats in response to nitrogen deposition have been detected, for instance, in montane heath in the Highlands of Scotland (Van der Wal *et al.* 2003), in UK grasslands (Stevens *et al.* 2004) and in UK habitats

as a whole (Maskell *et al.* 2010); such changes in vegetation alter the amount and quality of organic matter entering soil as plant litter and root exudates, which, in turn, affects the abundance and activity of soil organisms involved in nutrient cycling and the subsequent liberation of nutrients for plant growth (Bardgett & Wardle 2010). However, few, if any, changes in plant species composition and production have been observed in a series of long-term nitrogen addition experiments across the UK, suggesting than many terrestrial systems are no longer nitrogen limited and changes to both production and composition occurred during the early or mid-part of the 20th Century (Emmett 2007; RoTAP 2011). In many situations, increasing rates of recovery and restoration of the original plant community is now a major challenge for agri-environmental policy (Smith *et al.* 2008b).

Nitrogen enrichment may also have direct and indirect effects on soil biological communities that regulate plant nutrient supply and primary production including bacteria, saprophytic fungi (Donnison *et al.* 2000; Treseder 2008) and mycorrhizal fungi (Egerton-Warburton & Allen 2000; Frey *et al.* 2004). There is much uncertainty, however, about the relative effects on soil communities and primary production of acute large-dose fertiliser and chronic deposition of atmospheric nitrogen, and little is known about how responses vary between different habitat types. Ozone pollution can act against the promoting effect of nitrogen on plant productivity, which is an issue that is expected to increase; despite the decline of peak levels of ozone, background concentrations are increasing due to the continuing industrialisation of developing countries (RoTAP 2011).

There is also evidence that eutrophication can strongly impact on the phytoplankton community and fish populations of freshwater lakes (Moss *et al.* 2003; Moran *et al.* 2010). And in marine habitats, eutrophication can reduce and change marine biodiversity through mortality of fish, shellfish and invertebrates, which will impact on primary production and related ecosystem services (Chapter 12). It also encourages macro and microalgal blooms, which impact on primary production and may also harm fish through food chain effects, as discussed in Chapter 12. In shallow coastal and intertidal waters, the macroalgae can smother the soft sediments, impeding the flow of oxygen and nutrients to and from the sediment and affecting marine life and primary production (Chapter 12).

13.5.2.3 Climate change

Climate change will be a major driver of primary productivity in future years, with UK-level changes in temperature and precipitation regimes having potentially marked effects on vegetation productivity and composition of many broad habitats. For example, climate envelope models (which are based on species' environmental preferences) predict severe loss of biodiversity due to climate warming (Thuiller *et al.* 2006), with likely consequences for primary productivity. Moreover, climate change can cause shifts in species' distributions in terrestrial and aquatic habitats, causing changes in primary productivity. It can also affect species' phenology and interaction strengths, which can lead to mismatches in the life histories of consumers and their resources (Beaugrand *et al.* 2003; Post & Forchhammer 2008) and the decoupling of trophic interactions on which they rely (Visser & Both 2005; Memmott *et al.* 2007). As discussed for nutrient cycling, direct and indirect effects of climate change on soil microbial communities and mineralisation-immobilisation dynamics (via changes in carbon supply to soil) will influence plant nutrient supply; hence affecting the productivity of plant communities (Bardgett *et al.* 2008). Within individual habitats, drivers of change vary in their importance and can interact. For example, the potential enhancement of photosynthesis and primary production through elevated carbon dioxide will be limited by nitrogen availability in nitrogen limited and low-pollution areas (Hungate *et al.* 2003; Luo *et al.* 2004). Freshwaters are particularly vulnerable to climate change because: (i) many species within these fragmented habitats have limited abilities to disperse as the environment changes; (ii) water temperature and availability are climate-dependent; and (iii) many systems are already exposed to numerous anthropogenic stressors (Woodward *et al.* 2010). Likewise, effects of climate change on marine primary productivity are likely to be exacerbated due to ocean acidification, which is caused by increasing levels of carbon dioxide absorption by the oceans and the consequent changes to marine chemistry (Chapter 12).

In general, a large number of experiments, done under field and glasshouse conditions, highlight the complexity of responses to climate change and the need to include the effects of changing rainfall patterns, extreme events

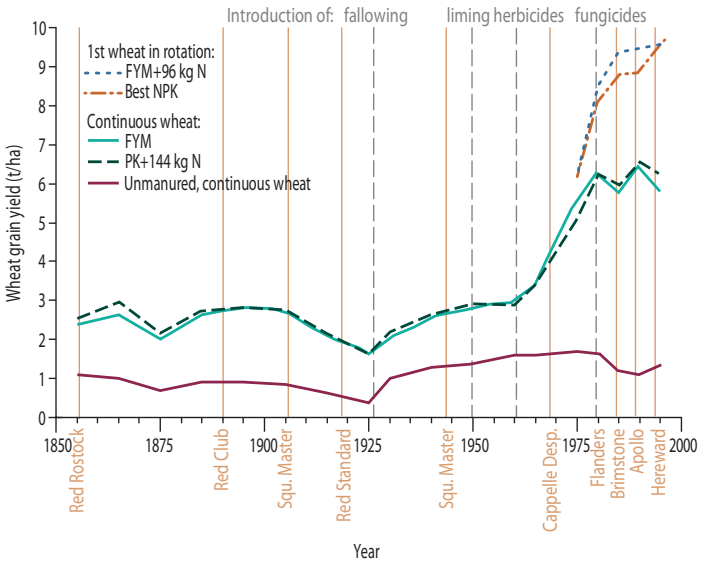

Figure 13.21 Yields of winter wheat (grain only, t/ha) on selected plots of the Broadbalk Wheat Experiment showing changes over time in varieties of wheat along the x-axis and changes in farming practice along the top of the graph. As a driver, this diagram illustrates the intensification of agriculture during the last few years which may be expected to continue. Less obviously, the increases in yield have been achieved by using more and more nutrients which have inevitably found their way into other ecosystems, terrestrial, freshwater and marine. Source: reprinted from Goulding *et al.* (2008).

and the interactive effects of different climate, pollution and management drivers. For example, a study of outdoor, shallow, freshwater mesocosms revealed that both warming and nutrient loading had a strong negative impact on sticklebacks, but combined, they resulted in extinction of this population (Moran *et al.* 2010). Studies of climate change impacts on primary productivity of heathland across a European climatic gradient also reveal contrasting effects of repeated summer drought on wet and dry sites; even within sites there are differential effects on plant species and nutrient cycles (Beier *et al.* 2008). In general, there is much uncertainty about the response of primary production to climate change and how responses vary across UK habitats, both terrestrial and aquatic, and in combination with other drivers of change. Future studies are needed which simultaneously vary two or more change drivers to determine their influence on primary production and other ecosystem services across a range of UK habitats. Such studies need to be linked to ecosystem models which provide an important tool for integrating this complex set of drivers on ecosystem processes (including primary production), and for making future projections about responses to climate change.

13.5.3 Consequences of Change

Primary production is the amount of carbon gained by vegetation, including new plant biomass, but also roots exudates and carbon transfer to symbionts. Therefore, any change in primary production will have significant consequences for all ecosystem services, including supporting services that depend on, or are affected by, the availability and rate of carbon accumulation (Smart *et al.* 2010). For instance, changes in primary production will have significant consequences for soil carbon sequestration in terrestrial habitats, given that the amount of carbon stored in soil is a function of carbon input by primary production and carbon loss through decomposition and plant respiration (De Deyn *et al.* 2008; Chapter 14). Changes in primary production also have far reaching consequences for terrestrial biodiversity conservation. Peak above-ground diversity commonly occurs at intermediate productivity (Al Mufti *et al.* 1977; Grime 1979; Grace 1999), with declining diversity at higher levels of productivity being due to competitive exclusion. Also, primary production acts as a major driver of below-ground communities and the processes that they drive, such as decomposition and nutrient cycling (Bardgett & Wardle 2010). Therefore, changes in primary production caused by drivers such as climate change and/or nitrogen enrichment are likely to have significant consequences for the biodiversity of above-ground and below-ground communities and the ecosystem services that they drive.

Primary production also regulates energy flow through food webs. Therefore, changes in primary production will cascade through both terrestrial and aquatic food webs, potentially altering their structure and function, their contribution to the delivery of ecosystem services (provisioning, regulating and cultural) and the goods and benefits that they provide. There is much uncertainty in this area, but one example of this concerns the North Sea where climate-driven changes in plankton communities have been shown to exert a strong control on cod stocks, complicating the management of this species (Beaugrand *et al.* 2003; Beaugrand & Kirby 2010). In terrestrial ecosystems, changes in primary production can affect herbivore populations and their consumers, but in turn, these changes can potentially modulate the effects of drivers such as climate change on primary production in the future (Bardgett & Wardle 2010). The Countryside Survey (2010) provides a capability to explore trade-offs and synergies between primary production and carbon sequestration which contribute to climate regulation and other intermediate and final ecosystem services such as water quality regulation and provision of habitat for wild species.

13.5.4 Options for Sustainable Management

Sustainable management of primary production in semi-natural and natural habitats, including rivers, lakes and oceans, is dependent on the maintenance of a complex set of physical, chemical and biological interactions that regulate the supply of growth-limiting resources like light, nutrients and water. Any disruption of this state, whether on land or water, will have knock-on effects for primary production and the regulating and provisioning services that they underpin. As highlighted above, a key driver of change in primary productivity in most habitats is nutrient enrichment (nitrogen and phosphorus). While this has reaped rewards in terms of provisioning services, for instance, increased crop production, it can have significant, detrimental effects on regulating services, such as water quality, and on the biological diversity that underpins the delivery of supporting, regulating and cultural services. Therefore, future management strategies need to be aimed at the better exploitation of these valuable resources to ensure efficient utilisation for provisioning services, whilst preventing transfers to aquatic systems or terrestrial habitats valued for their cultural, regulating or supporting services. Furthermore, given that long-term nutrient enrichment has already impacted significantly on UK vegetation production and contributed to observed declines in plant diversity, future management needs to be directed at reversing these effects through appropriate restoration management. For example, a key objective of UK agri-environment policy and Environmental Stewardship is the restoration of biodiversity in agricultural systems with potential benefits, albeit over long timescales, for other ecosystem services such as carbon storage and nutrient retention. However, restoration of habitats to deliver a full range of ecosystem services is very challenging. Research is needed to better understand the synergies between different ecosystem services (including primary production) and how management systems might be designed to reap multiple rewards for supporting, regulating and provisioning services.

Managing for the effects of climate change is problematic because impacts on primary production and related ecosystem services (e.g. nutrient cycling) are poorly understood and effects of climate change are often habitat-specific. In addition, whilst the role that primary production plays in the carbon cycle is well appreciated,

there is much uncertainty about the links between primary production and carbon cycling, especially in the soil. As a result, several gaps in understanding need to be tackled before sustainable management strategies aimed at climate change mitigation—for instance, through carbon sequestration on land and in waters—can be developed and implemented. Such linkages will also be affected by other drivers, such as nitrogen deposition; hence, future studies aimed at sustainable management of primary production and carbon sequestration need to consider responses to multiple, rather than just single, drivers of change.

13.5.5 Knowledge Gaps

As previously mentioned, data on the status and trends of primary production across UK habitats is limited. For terrestrial habitats, when data is available, it is mostly incomplete due to a lack of information about the amount of fixed carbon that is allocated below-ground to roots, which varies greatly across habitats (Jackson *et al.* 1996). This represents a significant gap in understanding, especially given the important roles that roots play in the delivery of terrestrial ecosystem services, including carbon sequestration, soil formation, nutrient cycling and the provisioning of food. There is also much uncertainty about the fate of land-derived fluxes of organic and inorganic carbon to surface waters, which can lead to major underestimations of primary production in aquatic systems. And there is little understanding of the effects of different land management regimes on soil carbon sequestration, despite them being proposed as effective means of mitigating greenhouse gas emissions. In inshore marine systems or intertidal areas, where water is shallow and visibility is often low due to suspended sediment, the contribution of macro and microalgae to seabed primary production is likely to be more significant. However, measurement of phytoplankton primary production via remote sensing in these areas is unreliable, and the extent of macro- and micro-algae in the UK is poorly mapped and quantified. This means the necessary data are missing to fully quantify UK marine primary production.

Another important gap concerns our understanding of how present or future changes in vegetation diversity and composition caused by land use and/or climate change, for example, influence primary production in UK habitats. Many experimental studies have explored how variations in species or functional group richness influence ecosystem processes, and several of these have found positive effects of diversity on primary production (Balvanera *et al.* 2006; Cardinale *et al.* 2006). But the interpretation and mechanistic basis underlying the results of these studies continues to be debated (Hooper *et al.* 2005). Indeed, many past diversity-function experiments have been based on randomly constructed plant assemblages that are artificial in nature and often immature, and while they have yielded a deeper mechanistic understanding of diversity-productivity relationships, their relevance to conservation management is debatable (Leps 2004; Bullock *et al.* 2007). Furthermore, the importance of species richness as a driver of productivity varies greatly among, and even within, studies (Fridley 2002; Hooper & Dukes 2004), and may be only minor in some habitats when compared to other biotic and abiotic factors (Grace *et al.* 2007). There is an urgent need for studies which explore the consequences of real scenarios of non-random changes in plant species diversity and composition on primary production and other related ecosystem services across UK habitats.

A related challenge concerns the role of soil biodiversity in regulating primary production. As discussed previously, soil biodiversity is known to be highly sensitive to a range of drivers (e.g. land use, climate change and nitrogen deposition), but the consequences of reductions in soil biodiversity for primary production, and the services that it underpins, are poorly understood (Bardgett & Wardle 2010). Another key challenge concerning primary production involves its role in the exchange of carbon dioxide with the atmosphere, and how NEE is controlled by abiotic, biotic and management factors (Wohlfart *et al.* 2008). In general, our understanding of how land use and other drivers (e.g. nitrogen deposition) influence NEE is limited, although evidence is emerging to suggest that factors such as grazing can trigger short-term shifts in NEE in grassland systems (Wohlfarht *et al.* 2008). For example, long-term grazing by sheep in a UK peatland has been shown to increase NEE and, hence, ecosystem carbon sink strength due to an increase in photosynthesis relative to respiration (Ward *et al.* 2007). Also, while year-to-year variation in NEE is mostly attributed to climatic variability, grazing has been shown to alter the impact of climate on land-atmosphere carbon fluxes (Polley *et al.* 2008). These findings indicate that predictive models need to accommodate biotic factors such as grazing in order to accurately simulate the dynamics of carbon dioxide fluxes in UK terrestrial ecosystems.

Finally, as also highlighted for nutrient cycling, it is becoming increasingly apparent that our ability to predict future responses of UK habitats to global change requires a greater understanding of the simultaneous effects of multiple global change drivers on primary production and other ecosystems services. There is much potential for interactions between global change drivers to amplify, suppress or even neutralise climate change-driven effects on the primary production and NEE, with likely consequences for the ability of UK habitats to provide ecosystem services. Studies in this area are still few and far between, and there is an urgent need for experiments which simultaneously vary two or more change drivers to determine their influence on primary production and other ecosystem services across a range of UK habitats.

13.6 Conclusions

Supporting services underpin the delivery of all other ecosystem services (i.e. regulating, provisioning and cultural services). Hence, understanding their response to key drivers such as climate change, land use and nitrogen enrichment is of fundamental importance for future sustainable management of the UK's land and water resource. In this Chapter, we have identified a number of key

trends in supporting services which have major implications for the delivery of other ecosystem services both now and in the future. First, while UK soils have taken thousands of years to develop (an average of <1 cm per century soil depth), there are concerns about the loss of soil by erosion under intensive agriculture, as well as the loss of organic matter from peat soils due to climate warming. Such soil loss has major implications for nutrient and carbon cycling in terrestrial and aquatic habitats, and for the delivery of regulating and provisioning services. Moreover, given that rates of soil loss can be much greater than formation, soils are essentially a non-renewable resource. Second, with regards to nutrient cycling, there is evidence of a widespread reduction in phosphorus available to plants in UK soils, which is possibly associated with increased primary production stimulated by increased atmospheric nitrogen deposition and climate warming. Also, nutrient cycles and primary production across all UK habitats have been, and continue to be, affected by anthropogenic inputs of nitrogen. In many UK habitats this has caused reductions in species diversity with uncertain implications for other ecosystem services such as carbon sequestration. In lakes, streams and coastal waters, it is well established that nutrient inputs of nitrogen and phosphorus from sewage, fertiliser runoff and soil erosion has caused major increases in primary production, again with uncertain implications for regulating and provisioning services. Warming has also caused changes in marine fauna, with potential knock-on effects for nutrient and carbon cycling, and ocean acidification could affect many marine organisms with uncertain consequences for supporting services. Finally, there have been few trends in the water cycle, but in the last 30 years, milder winters have been associated with increasing winter rainfall.

Although several key trends have been identified, our understanding of the mechanisms that underpin supporting services is limited, as is our knowledge of how these services will be affected by current and future drivers including climate change. It is also apparent that the mechanisms that underpin supporting services, such as plant production and nutrient cycling, are strongly context dependent. For example, studies indicate that the effect of plant species richness and community composition on ecosystem processes varies under different environmental conditions; likewise, drivers such as climate change, land use and nitrogen deposition will impact on supporting services differently across UK habitats. As a consequence, we identify many knowledge gaps concerning the need to better understand the mechanisms that underpin supporting services and their response to key drivers such as climate change. In addition, we highlight the urgent need for studies which explore the consequences of real scenarios of changes in biodiversity (i.e. species diversity and composition) on supporting services, and for experiments which simultaneously vary two or more key drivers of change to determine their influence on supporting and other ecosystem services across a range of UK habitats. Such studies are needed in order to develop sustainable options for the management of UK supporting services and the regulating, provisioning, and cultural services that they underpin.

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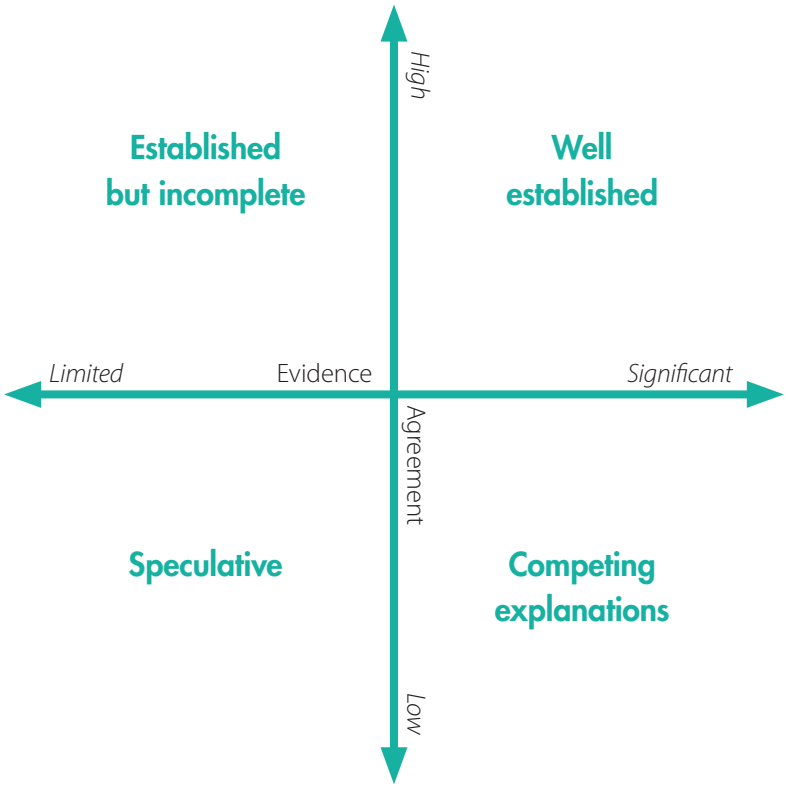
Appendix 13.1 Approach Used to Assign Certainty Terms to Chapter Key Findings

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

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

1. *Well established:* high agreement based on significant evidence
2. *Established but incomplete evidence:* high agreement based on limited evidence
3. *Competing explanations:* low agreement, albeit with significant evidence
4. *Speculative:* low agreement based on limited evidence



a. *Virtually certain:* >99% probability of occurrence
b. *Very likely:* >90% probability
c. *Likely:* >66% probability
d. *About as likely as not:* >33–66% probability
e. *Unlikely:* <33% probability
f. *Very unlikely:* <10% probability
g. *Exceptionally unlikely:* <1% probability

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