Summary

Human well-being is dependent upon renewable natural resources. Agricultural systems, for example, depend upon plant productivity, soil, the water cycle, the nitrogen, sulphur and phosphorus nutrient cycles and a stable climate. Renewable natural resources can be subject to biological and physical thresholds beyond which irreversible changes in benefit provision may occur. These are difficult to define and many are likely to be identified only once crossed. An environmental limit is usually interpreted as the point or range of conditions beyond which there is a significant risk of thresholds being exceeded and unacceptable changes occurring.[1]

Biodiversity loss, climate change and a range of other pressures are affecting renewable natural resources. If governments do not effectively monitor the use and degradation of natural resource systems in national account frameworks, the probability of costs arising from exploiting natural resources beyond environmental limits is not taken into account. Appropriate measurement methodologies need to be developed and validated to assess the capacity of natural resource systems to deliver benefits, such as relevant sets of indicators.

Decisions at local and regional scales need to reflect the implications and trade-offs for natural resource systems inherent in policy choices to determine possible consequences for current and future wellbeing. Valuation of changes in the benefits provided by natural resource systems are being incorporated into existing Cost Benefit Analysis (CBA) techniques used in policy impact assessment approaches. However, where there is a risk of thresholds being breached and potentially irreversible impacts occurring, additional policy safeguards to maintain natural resource systems within environmental limits are required.

Managing ecosystems to maximise one particular benefit, such as food provision, can result in declines in other benefits. The evidence base is not yet sufficient to determine the most effective ways to maintain benefit provision within environmental limits, but a range of policy responses are seeking to optimise multiple benefit provision, including:

- agri-environment schemes
- generic measures to enhance biodiversity, which may increase the capacity of natural resource systems to adapt to environmental change
- the use of ecological processes to increase overall natural system resilience to address problems such as flood risk management.

The consequences of large scale and potentially irreversible changes in benefits from natural resource systems, such as marine fisheries, could affect ecological security to such an extent that it is rational to minimise the risks, even if there is uncertainty as to exactly where the limits lie. However, the policy response to environmental risks to human wellbeing is mediated by the public response to that risk, with public acceptance affecting whether policies responding to a risk are enacted. There are significant challenges to successfully communicating environmental risks to the public.

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1 Introduction

1.1 Background

Changes in global environmental conditions have been brought about through a complex set of interactions between humans and the environment, as humans seek to satisfy their economic and social needs. Population growth, together with accelerated economic activities over the past two centuries have greatly increased natural resource use, as reflected in agriculture, fisheries, forestry, industry, transport, energy, urbanisation and the impacts of these activities such as pollution.

Human interaction with the global ecosystem has occurred through an economic system that has tended to under-value natural resource systems and the diverse range of goods and services that flow from them. Natural resources include land, water, air and associated living systems that comprise the mineral, plant and animal component of the biosphere, organised into ecological systems or ecosystems (Box 1). [2] They are the product of physical, chemical and biological processes over different time and spatial scales.[3]

Global environmental change due to increasing rates of natural resource use is reflected in multiple and interacting impacts such as climate change, biodiversity loss and dwindling water resources.[4] Although there have been significant periods of environmental change during previous geological eras, evidence from Antarctic ice cores suggests that global environmental processes have moved outside the range experienced over the last 700,000 years.[5] The Earth's environment has been unusually stable for the past 10,000 years, the Holocene era, during which temperatures, freshwater availability and biogeochemical flows were maintained within a relative narrow range beneficial to the human species.[6]

Since the Industrial Revolution, human activities facilitated by energy from fossil fuels began to drive global environmental change at a rate exceeding natural trends.[7] Since 1945, humans have changed ecosystems more rapidly and more extensively than in any comparable period of time in human history, with a global loss of some types of ecosystem, including 50% of wetlands, 40% of forests and 35% of mangroves.[8] The current decline in key environmental processes, such as climate regulation, is likely to foreshadow declines in other environmental processes that benefit humans.[9]

A number of reports in recent years, such as the Millennium Ecosystem Assessment (MA) and the series of "The Economics of Ecosystems and Biodiversity" (TEEB) reports, have all elaborated how the destruction of natural resource systems is posing a threat to human wellbeing in the near future.[10] How natural resources such as land, water, soil, plants and animals are used and managed affects human well-being for present and future generations. In the past, natural resource management has focussed narrowly on the scientific and technical understanding of natural resource systems and their ecology, usually to maximise yield of specific goods with a market value.

Box 1 Ecosystems

The UN Convention on Biological Diversity (CBD) and Millennium Ecosystem Assessment (MA) defined ecosystems as: "a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit". The key feature of ecosystems is that they are fully integrated systems,[2] with 'emergent properties' arising from interactions between the living and non-living elements of which they are composed.[11] Indicators of the boundaries of ecosystems are often defined in terms of the status of their dominant vegetation or environmental features, such as grassland or a lake, but are essentially a construct defined according to the scale of human interests and decision-making abilities. They can be defined as areas which share similar features in terms of:

- climatic conditions;
- geophysical conditions;
- dominant use by humans;
- surface cover (based on type of vegetative cover in terrestrial ecosystems or on fresh water, brackish water, or salt water in aquatic ecosystems);
- species composition; and,
- resource management systems and institutions (such as marine fisheries).

They usually have strong interactions amongst their components with weaker interactions occurring across the boundary of the systems. For example, the interactions between organisms in a lake are generally stronger than those between the lake organisms and those on surrounding land. Nonetheless, ecosystems do have porous boundaries, in the case of lakes, species such as toads, frogs and dragonflies will move between water and land in their life cycle, and there are also flows of water and minerals between land and water. As such, ecosystems are just one set of interactions nested within wider sets of interactions up to the global scale (and interactions with physical conditions beyond even this, such as solar radiation).[12]

Now, through policy approaches such as sustainable development, the management of natural resources encompasses wider economic and social dimensions such as the conservation of resources through more effective manufacturing processes, the reuse of materials sourced from natural systems, investing in natural capital and restoring and sustaining natural resources. Despite these policy changes, significant challenges remain in regulating the interaction between complex social and economic systems with a likewise complex planetary system.[13]

A key scientific concern has become whether there is sufficient understanding of these systems to achieve this against a background of rapid environmental change. This includes whether there are biological and physical thresholds or 'tipping points' beyond which exploitation of natural resource systems is unsustainable due to irreversible environmental change that is detrimental to ecosystem integrity and human wellbeing. Some commentators do not accept that crossing such thresholds will impact well-being and dispute the need for environmental limits. They argue that the profits from continuing economic growth will be used by future generations to reverse impacts on ecosystems or to substitute technology for goods and services arising from ecosystems.

However, there is evidence to suggest that political decisions need to be made now to regulate the interaction of economic systems and natural resource use to avoid human wellbeing being significantly impacted by 2030.[14] There is already evidence of environmental change affecting people's livelihoods globally. By 2030, the demand for food will increase by about 50%, water by 30% and energy by 50%, and the human population will have reached 8 billion. In addition, significant climate change adaptation and mitigation measures will need to be implemented in this timeframe.[15]

Recent arguments have been made that damaging levels of natural resource use cannot be decoupled from economic growth.[16] This report does not consider the full scope of interactions between humans and the environment or try to determine whether environmentally sustainable economic growth is achievable. However, policy areas that affect natural resource systems are increasingly seeking to consider complex trade-offs between social, economic and environmental objectives.[17]

Decision-makers could agree now the possible future states of the natural environment that are undesirable, and then consider alternative paths to avoid them. Future detrimental environmental change could be reduced by decreasing pressures on natural resource systems to increase their 'resilience'. However, the global degradation of natural resource systems has many drivers, including institutional and policy failure to control levels of exploitation. This report aims to summarise possible regulatory responses to increase the resilience of natural resource systems and interacting socio-economic systems.

Ecological Points of No Return – Chapter 2

By way of background, chapter 2 briefly summarises the concept of environmental limits, the importance of natural resource systems for human well-being and how human-induced pressures impact on these systems.

Accounting for Changes in Natural Capital – Chapter 3

Natural capital is usually interpreted as an economic metaphor for biological and physical resources available on the planet. The continued loss of natural capital through economic activities has created concerns that the critical amount of natural capital required to maintain human well-being is not being conserved. This chapter discusses the challenges relating to measurement of the impacts of economic growth on natural resource systems within national accounting frameworks.

Managing Human Interactions with Ecosystems – Chapter 4

This chapter describes how consideration of the use of and impacts on ecosystem structures and processes to maintain the flow of defined ecosystem services could provide a new focus for existing decision-making processes regulating impacts on natural resource systems.

Ensuring Resilience – Chapter 5

Resilience can be described as the amount of disturbance a natural resource system can absorb while providing the same level of benefits. This chapter considers what aspects of resilience can be meaningfully assessed and how uncertainties in relation to the impacts of human use of natural resource systems could be managed.

Environmental Risks and Limits – Chapter 6

This chapter considers the basis for how the risks arising from breaching environmental limits could be assessed in different policy areas, using food security as a case study where policy makers are seeking to address these issues.

2 Ecological Points of No Return

Overview

- An environmental limit is usually interpreted as the point or range of conditions beyond which there is a significant risk of abrupt irreversible, or difficult to reverse, changes to the benefits derived from natural resource systems that are judged to have an unacceptable level of impact on human wellbeing.[1]
- At a global level, the drivers of environmental change are continuing or increasing. Growing demands for natural resources have impacted the complex systems of plants, animals, and physical processes that sustain the flow of benefits from natural resource systems, which support the conditions necessary for life.
- An insufficient diversity of organisms in ecosystems to buffer environmental changes may result in 'ecological surprises' involving unexpected, irreversible, and negative alterations of key ecological processes. However, these shifts are difficult to predict and many such thresholds are likely to be identified only once breached.
- For most natural resource systems, environmental limits have not been defined, although possible environmental limits have been suggested at the global level for maintaining key biogeochemical processes.

2.1 Natural Resources

Natural resources include land, water, air and associated living systems comprising the mineral, plant and animal component of the biosphere (the part of the earth's crust, waters and atmosphere that supports life). These are organised into ecological systems, or ecosystems (Box 1). They influence, and in turn are influenced by, biogeochemical processes (the chemical, physical, geological, and biological processes and reactions that govern the composition of the natural environment) over different temporal and spatial scales that can be used in conjunction with other types of resources (financial, manufactured and social) to produce goods and services for human wellbeing. Natural resources are often referred to as natural capital (Section 3.1).[3]

The stock of renewable natural resources should be maintained over time, for example, a fish stock should not be harvested beyond sustainable limits or it will collapse. This report is concerned with renewable natural resources and related ecosystem services (NRES), which include land, water, air and associated living organisms from which goods and services beneficial to human wellbeing are derived (Annex A).

2.2 Environmental Limits

An environmental limit is regarded as the boundary beyond which exploitation of a natural resource poses increasing risks. A range of terms has been used in relation to environmental limits, listed in Table 1. They have been more specifically defined as the 'point or range of conditions beyond which the benefits derived from a natural resource system are judged unacceptable or insufficient'.[1] Environmental limits can be established on the basis of societal preference for the minimal acceptable output of benefits or the level of risk of crossing a biological or physical threshold at which unacceptable changes may occur (Chapter 6).

Environmental limits relate to the delivery of benefits, rather than the state or extent of any given ecosystem. For example, climate change may result in lower mean river flows in summer, leading to less dilution of treated effluent from sewage plants. This in turn can result in hypernutrification of watercourses (eutrophication), and the state of the watercourse deteriorating to the extent that the disposal of waste benefits are reduced to unacceptable levels. Once this water quality threshold is reached, the ability of the system to deliver other benefits, such as angling or recreational activities, may also not fall below acceptable levels. The environmental limit could be set in relation to the level of river flow, the concentration of sewage effluent or the impacts on angling or other recreational benefits.

Although biological and physical thresholds are objectively based on available evidence (Section 2.5), risk based limits reflect political considerations. There is scientific uncertainty about thresholds, and the consequence of exceeding them. This may result in a lack of political consensus about where environmental limits should be set (Section 6.1). Agreeing environmental limits also requires definition of unacceptable social and economic impacts arising from environmental degradation, a means of mitigating or reducing drivers/pressures of environmental change and a legislative framework within which they can be addressed, such as the UK Climate Change Act (Section

3.5). Several other terms are also closely associated with the concept of environmental limits, including environmental standards, targets and indicators.

Although there is often a lack of clarity in the use of these terms, the Department for Environment, Food and Rural Affairs (Defra) intends to publish a summary listing all existing UK statutory environmental limits as a means of ensuring that ministers are advised appropriately when policy proposals 'jeopardise' an environmental limit, such as air quality standards. A recent Government Economic Service review of the Economics of Sustainable Development stated the importance of the government sending clear signals to business about existing statutory environmental limits as an essential part of developing an environmentally sustainable economy.[17]

Limits, Standards, Targets and Indicators

Several terms are also closely associated with the concept of environmental limits, such as environmental standards, targets and indicators. [18] Environmental standards are generally used on a precautionary basis to inform target setting for environmental policies, such as those for reductions in levels of pollutants that affect human health. They include not only numerical and legally enforceable standards, but also those which are not mandatory, contained in guidelines, codes of practice or sets of criteria for deciding individual cases. They are judgements about the acceptability of environmental modifications resulting from human activities that are both:[19]

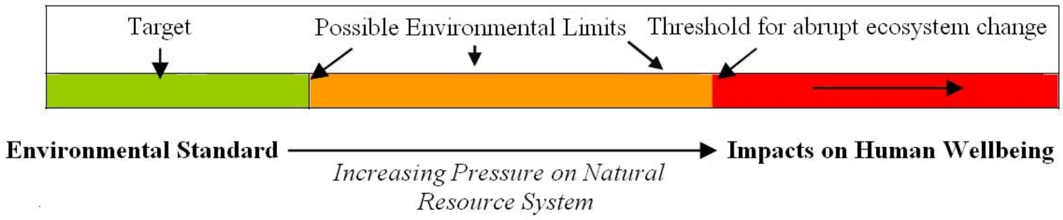
- formally stated after some consideration and intended to apply generally to a defined class of cases; and,
- expected, because of its relationship to certain sanctions, rewards or values to exert an influence, direct or indirect, on activities that affect the environment.

Environmental standards seek to reduce pressure on natural resource systems and avoid limits being exceeded. For example, the 'good ecological status' target of the Water Framework Directive is based on compliance with more than 50 standards set on a water-body specific basis (rivers, lakes, canals, estuaries, coastal waters and ground waters) relating to biological, chemical and physical quality, with environmental quality standards for levels of specific pollutants, such as pesticides (POSTnote 320).

Table 1: Terms Used in Relation to Environmental Limits

Term	Definition
Thresholds, Tipping Points and Regime Shift	An ecological or biophysical threshold is the point at which there is an abrupt change in the properties of an ecosystem or ecosystem processes in response to pressure from human activities or other drivers of environmental change. Shifts typically result from a combination of gradual changes in drivers, such as land use change, that initially appear to have little or no apparent impact up to the threshold, until an external shock such as storm, fire or disease outbreak, causes the threshold to be crossed. Some thresholds are reversible transitions, others may be extremely difficult or impossible to reverse (Section 2.7).
Critical Natural Capital	The level of unexploited natural resources required to maintain the capacity of ecosystems to carry out processes important to human wellbeing at acceptable levels (Section 3.2).
Critical Loads and Levels	The amount of "pollutant" that an ecosystem can absorb before there is a change in the natural resource system and/or in a particular ecological process. For example, critical loads are used to specify the maximum rates of annual deposition of oxides of sulphur and nitrogen emissions permissible while avoiding adverse effects on soils and/or freshwater systems (Annex 1).
Limit Reference Point	The level of use or pressure at which the capacity of a renewable natural resource system to sustain itself is damaged, for example, the maximum values of fishing mortality that should not be exceeded, or minimum values of the biomass of fish to be maintained, to avoid collapse of a fish stock.
Carrying or Assimilative Capacity	The level of use of a natural resource system beyond which undesirable changes will happen to the system, for example, the size of a species population that can be sustained by a particular ecosystem. The carrying capacity of ecosystems to support human populations is the basis for calculating 'ecological footprints' (Section 3.4 and Annex 2).
Safe Minimum Standards	Used within the context of economic analysis as the point at which it becomes unacceptable to trade-off environmental losses against economic gains because unacceptable changes will occur in natural resource systems. In cases of uncertainty, there is a requirement to err on the side of caution, with a caveat of unacceptable costs, for example, in the Water Framework Directive (Section 4.10)

Figure 1 The Relationship between Targets, Precautionary and Environmental Limits.[1]



Similarly, targets and indicators are used to monitor and set progress towards environmental standards (Section 3.7). If ecological thresholds are breached, impacts on human wellbeing could form the basis of legal action. However, there is a lack of clarity in the relationship between limits and existing environmental standards, targets and indicators (Figure 1). This is a reflection of uncertainties about where the ecological thresholds for many natural resource systems lie, and what would be the consequence of exceedance for human wellbeing.

Applying Environmental Limits

The term 'environmental limits' has been used in a number of different contexts, most commonly in terms of the consumption of products and services and the ability of natural resource systems to sustain this (Chapter 3), and in the management of impacts on natural resource systems and ecosystem services (Chapter 4):

- The former of these is usually referred to as 'sustainable consumption and production'. The main focus of this policy area is to promote better products and services, which have lower environmental impacts from the use of energy, resources, or hazardous substances through cleaner, more efficient, production processes and reducing impacts throughout the rest of the lifecycle of a product or services. The aim of these policies is to 'decouple' economic growth from environmental impacts.
- The latter policy area is usually referred to as the 'ecosystem approach' – "a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way".[20] The ecosystem approach provides a framework for considering impacts on natural resource systems as a whole and maintaining options for future uses. In the context of regulatory decision making, this report mainly considers how the concept of environmental limits could be applied through the ecosystem approach to the delivery of benefits derived from particular ecosystem services.

The ecosystem approach includes consideration of the ecological thresholds at which changes occur, as well as societal trade-offs between different benefits. Ecological thresholds exist at a range of scales from the global to the national, and local. Although the concept of environmental limits tends to place emphasis on thresholds, the capacity of natural resource systems to recover from impacts or pressure is also integral. Some natural resource systems become more vulnerable to irreversible changes if the resilience of the system is reduced by a plurality of pressures (Chapter 5). The following sections of this chapter summarise the drivers of change in natural resource systems, the effects of change on natural resource systems and increased risk of breaching of ecological thresholds.

2.3 Drivers of Change in Natural Resources

Human-induced causes of environmental change in natural resources are actions that directly or indirectly destroy natural systems or reduce their quality. Based on various indicators, such as human population density, numbers of settlements, roads and agricultural activity, around 83% of the global terrestrial biosphere has been classified as being under direct human influence.[21]

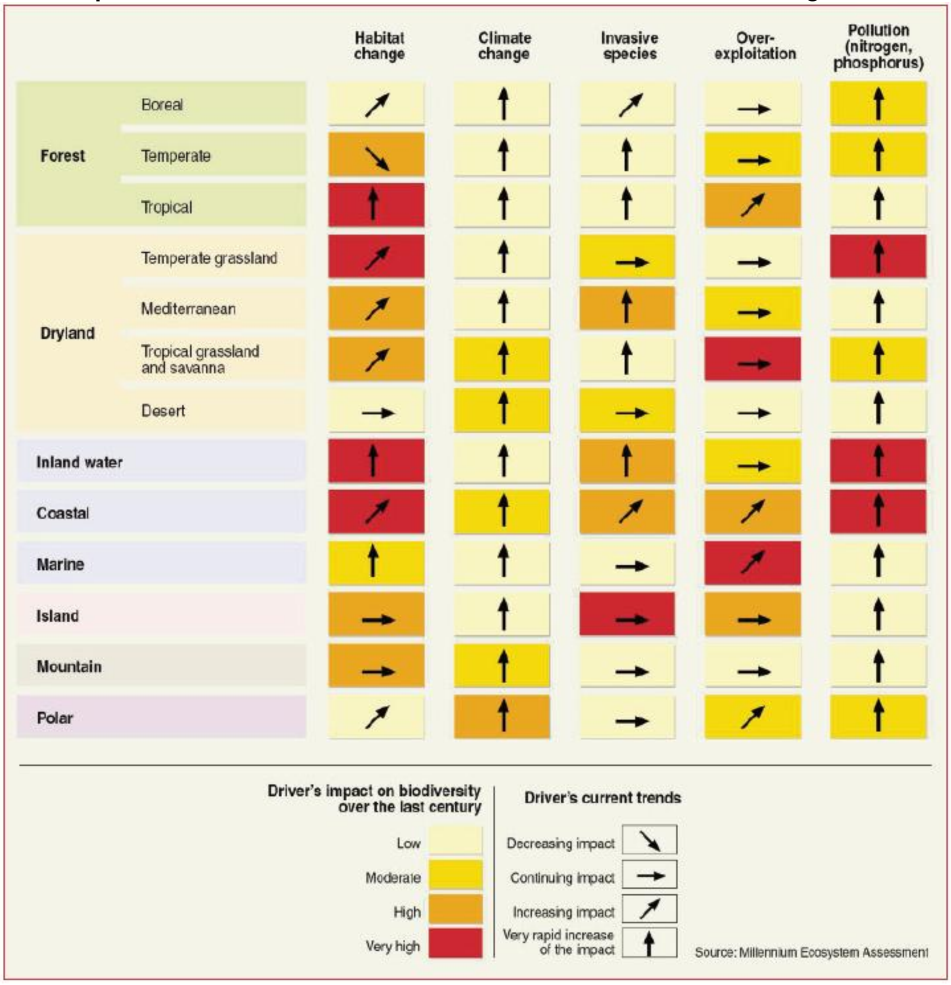
Approximately a quarter of the Earth's terrestrial surface has been converted to cultivated systems.[22] The recent report on The Economics of Ecosystems and Biodiversity (TEEB), commissioned by the G8 + 1, estimated that, including marine areas, at least one-third of the area of all biomes is significantly affected by human activities.[23] A biome is a regional ecosystem characterized by distinct types of vegetation, animals, and microbes that have developed under specific environmental conditions, such as grassland, forests or desert.

Figure 2 shows the main direct drivers of global change over the last 50-100 years in different biomes, along with the current trends of those drivers. Indirect drivers are broad-scale influences that affect natural resource systems by changing overarching environmental conditions or the way society interacts with natural resource systems, such as the dietary shift in developing countries to a greater consumption of meat and dairy products. The principal drivers of change in natural resource systems in Figure 2 are either constant or increasing on these timescales.

Although the intensification of land use change is a fundamental driver of environmental change, it is not the sole cause, and specific mechanisms of environmental change differ according to biome, geography, climate, economic context of regions, trade patterns and governance structures. These drivers are diverse, varying according to ecosystem characteristics, but are known to include:

- habitat loss and fragmentation;
- agricultural production (including soil degradation, commodities and nutrient loading);
- overexploitation of marine fisheries; climate change (including ocean acidification);
- water use (including abstraction and wetland drainage);

Figure 2 MA Representation of the Main Direct Drivers of Environmental Change in Different Biomes.



The cell colour indicates impact of each driver on the given biome over the past 50-100 years. High impact means that over the last century the particular driver has significantly altered biodiversity and ecosystems in that biome. The arrows represent the trend in drivers, diagonal and vertical arrows indicate increasing trends in impacts, horizontal arrows a continuation in current trends.

- urbanisation;
- infectious disease and invasive species;
- pollution and waste disposal;
- technological change and human consumption of natural resources; and,
- economic and population growth.

The relative importance of different pressures as drivers of environmental change is incompletely understood, specifically with regard to how these different stressors interact to affect the individual components of natural resource systems. However, natural habitats in most parts of the world continue to decline in extent and integrity, despite progress in recent decades in slowing the rate of loss for some habitats in some regions, such as tropical rainforest in Brazil. In particular, freshwater wetlands, sea ice habitats, salt marshes, coral reefs, sea grass beds and shellfish reefs are showing serious declines.[24] The trends for the drivers shown in Figure 2 pose a significant constraint to achieving the 7th Millennium Development Goal of ‘ensuring environmental sustainability’ that has a number of targets against which progress is being measured, including:

- target 7A: Integrate the principles of sustainable development into country policies and programmes; reverse loss of environmental resources;
- target 7B: Reduce biodiversity loss, achieving, by 2010, a significant reduction in the rate of loss as reflected in: proportion of land area covered by forest; CO_2 emissions, total, per capita; consumption of ozone-depleting substances; proportion of fish stocks within safe biological limits; proportion of total water resources used; proportion of terrestrial and marine areas protected and proportion of species threatened with extinction.

The Strategic Plan of the convention on Biological Diversity (CBD) or the "Aichi Target" was adopted at Nagoya in October 2010. The target includes a commitment to halve, and where feasible, bring close to zero, the loss of natural habitats and also to protect 17% of terrestrial and inland water areas and 10% of marine areas. Also included are measures to control invasive species and a protocol on access to genetic resources and the fair and equitable sharing of benefits arising from their utilisation.[25]

In response to increasing concerns, an Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) has also been created under the auspices of United Nations Environment Programme (UNEP). Similar to the Intergovernmental Panel on Climate Change (IPCC), IPBES will be responsible for the provisioning of authoritative, independent, credible, inclusive, internationally peer-reviewed and policy-relevant scientific advice on changes in biodiversity and ecosystem services and their implications for human well-being at multiple scales.[26]

The MA provided a snapshot of pressures on ecosystems over the last fifty to a hundred years. The state of ecosystems locally may differ from the overall global perspective and over shorter timescales than used in the assessment. For example, agri-environment policies have changed markedly in the last 10-15 years in EU countries and have been shown to have beneficial effects. In 2001, the EU set out to halt biodiversity loss in the EU by 2010. Although it did not meet this target, it has agreed a new target to:

"*halt the loss of biodiversity and the degradation of ecosystem services in the EU by 2020, restore them in so far as feasible, while stepping up the EU contribution to averting global biodiversity loss."*

The EU also recognised:

"*the need to keep human activity within safe ecological limits and to avoid human-induced loss of biodiversity through extinctions and passing other ecological points of no return*."

2.4 Role of Biodiversity in Ecosystem Services

Biodiversity is the number, abundance, composition, spatial distribution, population structure and interactions of living organisms and the physical habitats in which they are found.[13] Interactions between the components of biodiversity give rise to ecological processes that contribute to 'ecosystem services', such as food, soil stabilisation, flood regulation, regulation of the chemical composition of the atmosphere and pollination (Box 2), that human wellbeing directly depends on.

All components of biodiversity, from the genetic diversity within a species to the spatial arrangement of habitats, can play a role in these interactions (Figure 3).[13] This flow of services includes acting as a source of materials, a sink for wastes produced, regulation of air, climate and water, production of food and fibre, recreation, and aesthetic and cultural values of nature.[27] Human wellbeing, while buffered against environmental changes by culture and technology, is fundamentally dependent on the flow of ecosystem services.

The MA separated these services into four categories: *provisioning services*, such as food and water; *regulatory services*, such as flood and disease control; *cultural services*, such as spiritual and recreational benefits; and *supporting services*, such as soil formation, photosynthesis and nutrient cycling that maintain the conditions for life on Earth.

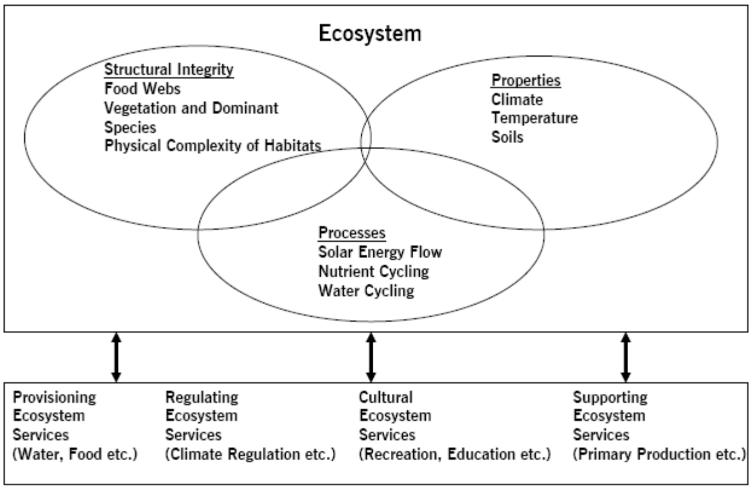
Determining the Impact of Biodiversity Loss on Ecosystem Services

Measuring ecosystem services, understanding the interdependencies between drivers of environmental change and ecosystem services and valuing ecosystem services are considered a key scientific challenge.[28] The need for good quality evidence to contribute to policy making and sound decisions in the area of ecosystem services is outlined in the recent "Defra's Evidence Investment Strategy".[29]

The level of ecosystem services supported is linked to the quality and quantity of the ecosystems that support those services. There are numerous different stable ecosystem states possible for a given location, each with different combinations of levels of ecosystem service delivery. For example, both forest and moorland ecosystems can be sustained in UK upland areas, and choices between these alternate states could be made through agri-environment schemes to achieve desired service levels, such as water quality and carbon sequestration.

Ecosystem services may be provided by individual species, groups of species with similar traits, or the interactions of entire ecological communities. However, links between measurable properties of ecosystems and ecosystem services are often difficult to quantify (Section 3.6 and Annex B). Understanding of which critical ecological elements underpin the long term provision of most services, how ecosystems function and how biodiversity influences this functioning, is incomplete.[30]

Figure 3 Interactions Between the Components of Ecosystems, Living Organisms and the Environment in Which They Live, Give Rise to Ecosystem Services From Which Humans Benefit.[31]



Box 2 Pollination Services

About 80% of British plant species[32], including many crops, make use of insects to transfer pollen between flowers to produce seeds and fruits. Without their pollinating insects, these plants would reproduce less well, or not at all. This could resonate through ecosystems, for example, affecting birds whose diet includes seeds and fruits from such plants. Pollination is therefore an essential ecosystem service which maintains biodiversity and supports other vital ecosystem functions. Pollination services are a public good, and many pollinators are charismatic insects that generate strong public interest (contributing to cultural services, Figure 3). There are thousands of insect species in the UK which may contribute to pollination, including bees, hoverflies, butterflies and moths. Pollinators may be generalists, pollinating many species effectively, or specialists that visit only a few species. Different flower types have different pollinators. Several characteristics of bees, such as their size, hairiness and foraging behaviour, indicate they pollinate flowers more efficiently than other insects (POSTnote 347).

Pollination has a direct economic value through increasing the yield and quality of insect-dependent crops. In the UK, this includes oilseed rape, orchard fruit, soft fruit and beans. Total loss of pollinators could cost up to £430m a year, about 13% of UK income from farming[33]. Of these pollinators, domesticated honeybees are responsible for a maximum of 34% of crop pollination (on the basis of the number of hives in the UK and their proximity to crops), with the remainder carried out by wild pollinators. Insect-dependent crops can be pollinated by hand, but the cost of this would be prohibitive (initial estimate of labour costs is £1500m a year). [34] Pollination, through its essential role in maintaining biodiversity and ecosystem functioning, also provides indirect benefits to agriculture. These indirect benefits have not been valued, but probably exceed the value of direct benefits. Requirement for pollinators may increase in the future as:

- demand for insect-pollinated crops is increasing; the area cultivated has risen by 38% since 1989 in the UK; and,
- demand for food and biofuel is increasing, placing more pressure on land and yields.

The overall abundance of pollinators has probably decreased since the 1970s, with some species undergoing dramatic declines. Farming intensification has reduced the availability of food plants and nesting sites through conversion of semi-natural land to intensive farmland and changes in agricultural practices. These affect wild pollinators more because they are totally reliant on resources available in the landscape; the number of visits to crop fields by wild pollinators tends to drop with distance from semi-natural areas[35]. Therefore, effective pollination by wild pollinators requires cropland to be interspersed with semi-natural areas. The number and diversity of pollinators necessary to maintain pollination services is not known, so it is hard to predict when serious consequences will arise due to pollinator loss. The effect of losing pollinator species can be explored through computer models of plant-pollinator networks. These suggest that networks are fairly robust to the removal of specialist pollinators; plant species diversity is maintained until 70% of pollinator species are lost, provided the most specialist are lost first[36]. Losing generalist pollinators, like honeybees, has more severe consequences for diversity.

As a supplement to the conservation of semi-natural habitats, sowing wildflower seed mixes is a quick and relatively cheap way of creating habitats that benefit pollinators. Field trials run by the Centre for Ecology and Hydrology found that bumblebee abundance was 14 times higher in wildflower margins than in a conventionally managed cereal crop. Pollinator abundance was 12 times higher in farmed landscapes that were managed using targeted agri-environment schemes for pollinators and other wildlife, compared with areas outside the schemes (standard cross-compliance).[37] These trials indicate that maximum benefit to pollinators from sowing seed mixes can be achieved by:

- sowing of more diverse mixtures that provide pollen and nectar throughout the flowering season and are long-lasting;
- a landscape-scale approach which provides connectivity between flower-rich patches and enough habitat to sustain viable populations of pollinators; and,
- targeting habitat creation to the most intensively farmed landscapes.

The area of pollinator habitat needed in the landscape is not known, but expert opinions range from 1.25% to 2.5%. Estimated costs of creating these wildflower meadows vary depending on whether compensation is paid for income foregone through Environmental Stewardship (ES) agri-environment schemes (section 5.8), which cost £50-100m a year, or whether seed costs alone are paid, at £3.3-6.7m per year. ES could provide a framework to deliver more pollinator habitat. However, uptake of pollinator-targeted options in ES is low, covering about 0.05% of English agricultural land. In addition to creating wildflower meadows, promoting the inclusion of clover leys in crop rotations and restoring habitats like woodlands, hedgerows and grassland is advantageous for pollinators. These measures would also provide a range of additional ecosystem services, beyond pollination services.

The role of many species in ecosystem services is either not or poorly understood and may not become apparent until after they are lost from an ecosystem. In some cases, a few dominant species drive key ecological processes and the wider diversity of species is critical only when the system is disturbed or subject to environmental change. For example, rainfall declines could lead to drought resistant plant species, and organisms that feed on them, playing a more prominent role in ecological processes in an ecosystem. In other cases, such as processes which depend on micro-organisms, they can depend upon a wide diversity of species (Annex A).

Different species fulfil different functions within ecosystems, with key species or groups of species that perform a particular ecological role in an ecosystem likely to play a major role in service provision (Box 3). Functional diversity, the component of biodiversity that concerns the range of things that organisms do in communities and ecosystems, is important in maintaining processes in ecosystems (Figure 3).[38]

There is good evidence that improvements in some ecosystem processes are associated with increased numbers species (for example plant productivity, Box 4). In response to changing environmental conditions, different species may fulfil different roles with increased biodiversity providing an insurance function (Section 5.2). Many ecosystem services are underpinned by more than one ecological process, for example flood regulation and food production.

The components of ecosystems that that are known to provide an ecosystem service are usually referred to as 'Ecosystem Service Providers' (ESP), and the species or groups of species that perform functions necessary to deliver an ecosystem service are referred to as 'Service Providing Units' (SPU).[39]

However, it should be noted that the species providing beneficial services are supported by direct and indirect interactions occurring between the biodiversity components of the ecosystem, without which service providing species would not be available (Annex A).[40] Ecosystem functions, ecological processes that maintain the integrity of ecosystems, often rely on interactions between organisms at different positions in the food chain.

For example, pollination (Box 2) is affected by interactions between the diversity of plants and other organisms, such as soil organisms, herbivore grazing, the diversity of pollinators and the organisms they interact with, such as predators.[41] The greater the number of interactions in an ecological process, the harder it is to understand the role played by the individual components of biodiversity within that process.

Determining the risks arising from biodiversity loss for ecosystem service provision requires both monitoring changes in the quantity and quality of ecosystems and greater understanding of how different organisms support ecosystem service provision within ecosystems. Assessments of pressures impacting on ecosystems can reveal the nature of changes in the states of ecosystems and the delivery of ecosystem services that stem from those ecosystem states.

This would allow policy responses to the drivers of pressures that change ecosystem service delivery to be informed by an appropriate evidence base. However, research at a scale that can incorporate the inherent complexity of both natural and managed ecosystems will be required before the characteristics of biodiversity needed to provide ecosystem services can be defined with any certainty.[30]

Box 3 Examples of the Functions of Species in Beneficial Services

Eurasian jays and oak seed dispersal:[42] In the oak forest of the Royal National Urban Park in Stockholm, critical seed dispersal is provided by Eurasian jays. The park provides important cultural and recreational services, with over 15 million visits a year. The oak forest is a key component of the park and oak trees play important roles in other ecological processes in the region, such as acting as habitats for an array of insect species. The foraging and dispersal behaviour of the jays facilitates acorn germination to an extent much greater than that of any other animal species in the park. A study estimated the replacement cost of the seed dispersal provided by jays in terms of the cost in dollars of human labour to seed or plant the oak trees. An average of 33,148 oak saplings a year are required to maintain the forest, which could be achieved by approximately 12 resident jay pairs, but this is a low estimate which does not allow for the impacts of environmental change on the jay population. The estimated costs were $4,900 for seeding or $22,500 for replacing the services of a single pair of jays. The current population is estimated at 84 individuals, although breeding pairs require conifer forest in a nearby location for nesting for part of the year.

Great tits and pest control:[43] Great tits can be encouraged reduce caterpillar damage to apple orchard crops substantially. At a density of one to six breeding pairs of birds per 2 hectares (ha), caterpillar damage is reduced by up to 50% compared with control sites with no breeding pairs. The density of breeding pairs is critical, as caterpillars are an important part of the bird's diet during the breeding period and are a preferred food item for nestlings. For the great tits to provide a pest control benefit, the breeding season needs to be synchronised with peak caterpillar activity and the relevant stage of crop development, as well as to have sufficient numbers of breeding pairs in the location of the orchard. The need for this benefit will increase with requirements for growers to reduce pesticide usage, caused by legislation and consumer demand, and the service can be facilitated by growers through the provision of nest boxes in orchards.

Much of the existing evidence base reflects narrowly-defined research on individual components of biodiversity without a wider understanding of the interrelationships.

2.5 Declines in Biodiversity

Humans are modifying both the types and numbers of species in ecosystems on a global scale, although the impacts of such changes on ecosystem processes remain uncertain. The economic and market failures that drive biodiversity loss have been extensively reviewed by the recent TEEB report and other studies. [44] These all suggest that there has been insufficient integration of the value of biodiversity and ecosystem system services into government policy frameworks, strategies and programmes to address the drivers of biodiversity loss.

Loss of biodiversity and a decline in ecosystem services are expected to continue at an increasing pace in the coming decades as the economic, demographic and market forces underlying the drivers of environmental change are unlikely to be addressed in the short term. The UN 2010 biodiversity target to halt biodiversity loss and the accompanying twenty one sub-targets have not been met at the global level. The 2010 UN Convention on Biological Diversity report, "Global Biodiversity Outlook 3" stated that:[45]
"*Despite an increase in conservation efforts, the state of biodiversity continues to decline, according to most indicators, largely because the pressures on biodiversity continue to increase. There is no indication of a significant reduction in pressures upon it*"

Increasing demands for provision of food, fresh water, energy, and materials have impacted the complex systems of plants, animals, and biological processes that sustain ecosystem services. One indicator of human pressure on natural resource systems is how much of the energy captured by primary production is cumulatively appropriated by human activities, reducing that available for other ecological processes and outcomes, thereby altering the composition of the atmosphere, levels of biodiversity and energy flows within food webs. [46] On measures of declines in primary productivity, the Global Analysis of Land Degradation and Improvement estimated that 24% of the world's land area is undergoing degradation including around 30% of all forests, 20% of cultivated areas and 10% of grasslands.[47]

Studies to develop a global and regional indicator to estimate this human appropriation of net primary production (see HANPP, summarised in Box 4), suggest that a global average of between 24 to 32% of net primary productivity is appropriated for human use. This substantial proportion is illustrative of how material flows from human activities have become a major component of the earth's biogeochemical cycles.

Biodiversity loss can simply mean the reduction in the numbers of plant and animal species, but it usually encompasses all the components of biodiversity, ranging from the loss of genetic diversity within species to physical habitats, and the loss of ecological processes within ecosystems. These types of damage to biodiversity increase dramatically in the course of economic development until, at a certain level of wealth, opportunities for biodiversity conservation can potentially be improved, albeit applied to an already-impoverished natural environment.

New investment or increased international demand in particular economic sectors, can be linked to habitat destruction, resource depletion and industrial pollution.[23] Global biodiversity is projected to further decrease from about 70% in 2000 to 63% in 2050 (compared with natural intact ecosystems).[48] Recent estimates place the loss of biodiversity worldwide a cost equivalent to 7% of world GDP in 2050.[49]

Species extinction is a normal part of ecological and evolutionary processes. However, the current rate of extinction is estimated to be 100 to 1,000 times higher than it would be without the impacts of human activities:

- Natural habitats and populations of species are declining by an average of 0.5 - 1% per annum, with the majority of loss occurring in the developing world.[50] For terrestrial ecosystems, habitat loss is largely accounted for by conversion of wild lands to agriculture, which now accounts for about 30% of land globally. The IUCN listed 21% of all known mammals, 30% of amphibians, 70% of plants and 35% of invertebrates as 'threatened' with extinction if current conditions persist. Species are classified by the IUCN as extinct, extinct in the wild, critically endangered, endangered, vulnerable, near threatened, least concern and data deficient. Those species that are classified as critically endangered, endangered or vulnerable are considered as 'threatened'.[51]
- The population of wild vertebrates fell by an average of nearly one third (31%) globally between 1970 and 2006, with the decline especially severe in the tropics (59%) and in freshwater ecosystems (41%). 42% of all

Box 4 Human Appropriation of Net Primary Production (HANPP) in Earth's Terrestrial Ecosystems

All life on earth is directly or indirectly reliant on primary production: the production of organic molecules with their associated embedded energy from carbon dioxide and water through the process of photosynthesis by living organisms. The main source of energy captured by photosynthesis is sunlight. A number of micro-organisms (lithotrophs) are able to utilise the energy from inorganic molecules, such as sulphur compounds, ferrous iron or ammonia which, although significant for biogeochemical processes, contribute negligible amounts of energy to ecosystems overall.

The bodies of living organisms within a unit area make up a standing live biomass, biomass being the mass of organisms per unit of area per year, which ecological scientists estimate as mass of carbon per unit area per year (g $C/m^2/yr$). Most of the biomass of an area will consist of plant species, the primary producers. In terrestrial ecosystems, the vast majority of primary production is by vascular plants, whereas in marine ecosystems the dominant primary producers are algae, the most significant of which are single-celled phytoplankton. Gross Primary Production (GPP) is the rate at which a given amount of chemical energy is captured and stored as biomass. Net Primary Production (NPP) is the fraction of this energy not used by the producing organism for respiration or cellular maintenance and is the organic matter available to the organisms that consume the primary producers. This amount of organic matter represents the primary food energy source for the world's ecosystems.

How HANPP is calculated depends on how it is defined, and a previous lack of standardisation has resulted in a variation in the figures derived. HANPP is calculated by determining the potential NPP of the vegetation that would be present in the absence of human activity, the actual NPP currently present with human activity and the amount of NPP harvested by humans. It is most informative when done on a spatially explicit basis by incorporating geographic information system (GIS) technology. The vegetation that would be expected in the absence of land use change can be determined through use of a Dynamic Global Vegetation Model. Datasets on agriculture and forestry are available from a number of sources, such as the Food and Agriculture Organisation (FAO), and estimates of grazed biomass can be extrapolated from livestock statistics, but statistics on the vegetative cover for urban and wilderness areas remain a major challenge.

HANPP varies spatially from almost zero to many times the local primary production, indicating the degree to which human populations depend on net primary production 'imports'. It is estimated that, on a regional scale, some areas such as Europe and south central Asia, consume more than 70% of their regional NPP, with other regions consuming less than 15% of NPP. At the local scale, consumption of NPP can vary from nearly 0% in sparsely populated areas to over 30,000% in large urban centres. International trade means that environmental impacts of humans consuming NPP are realised away from where the products are consumed, and understanding the flows of NPP-based goods is a critical challenge to determine the global impacts of human populations. The average per capita HANPP for industrialised countries is almost double that of developing nations, which constitute 83% of the global population. If consumption of HANPP in developing countries were to reach that of developed nations, it would significantly impact on ecosystems in those countries and elsewhere through increased NPP imports. However, it is also notable that developed countries process NPP more efficiently through improved technologies that reduce waste.[52]

amphibian species and 40% of bird species are declining in population.[24]

- Freshwater species such as fish, frogs, turtles and crocodiles are some of the worst affected, becoming extinct at six times the rate of marine and terrestrial species.[53]
- Species in all groups with known trends are, on average, are closer to extinction than previous years, with amphibians and warm-water reef-building corals showing the most rapid declines in status.[24]
- Many ecosystem services such as decomposition and nutrient cycling are dependent upon micro organisms, such as mychorrhizal fungi. Little is known about the changing status of these organisms, creating difficulties in determining ecosystem trends.[54]

Biodiversity Decline in the UK

Biodiversity indicators suggest a widespread decline of species populations in nearly all habitats in Europe, with severe declines in farmlands - an average of 23% between 1970 and 2000.[55] Large declines in agricultural landscapes of populations of pollinating insects, such as bees and butterflies, and birds, which disperse seeds and control pests, may have consequences not only for agricultural production but also on maintaining species diversity in natural and semi-natural habitats (Box 2). Declines in the biodiversity of the UK have been extensively reported in other recent publications,[56] as highlighted in the recent "Making Space for Nature" review carried out for Defra.[57]

England alone supports at least 55,000 species (not including micro-organisms), and includes biodiversity of global and European importance, including 18% of the world's heathland and more chalk rivers than any other country. However, there was a significant loss of biodiversity in England during the last century, with only 3% of species-rich grassland habitat that existed in the 1930s surviving to 2000. There is also continuing loss of important habitats, such as lowland hay meadows, that have not been protected by conservation designations. By 1980, over a quarter of upland heathland had been lost in England, with losses of 36% in Cumbria. Widespread declines in the condition of the remaining habitat still continue.[58]

Although environmental policy is devolved in the UK, biodiversity policy is co-ordinated through the UK Biodiversity Action Plan (BAP). Of the 45 BAP habitats in the UK, trend data for 2008 suggest that 42% are declining to some degree, 20% are stable and 20% are increasing to some extent (18% no clear trend/unknown).[59] The UK Joint Nature Conservancy Council (JNCC) has identified that at least 11 of these BAP habitats were declining as result of agricultural practice.[60] However, reasons for overall biodiversity declines are not due solely to agricultural practice but to a range of complex interacting factors, including climate.

The 2002 "Changing Flora of the UK" report showed a significant rise in non-native species, together with a decline in species intolerant of high soil fertility. [61] Changes in the composition of many vegetation communities have been driven by increased nitrogen and phosphorus inputs into agricultural soils which also affect soil faunal biodiversity (a switch from fungi-dominated nutrient communities to bacteria-dominated communities) and the composition of above ground ecological communities.[62] Nitrogen additions to farmland increased by over 300% from 1957 to peak in the 1980s, but decreased significantly by 2007. There have been continued declines in monitored groups of species used as biodiversity indicators in England in the past decade[63], including:

- The 30 species of birds monitored, in routine breeding bird surveys by the British Trust for Ornithology, have average decline of 54%.
- Monitored butterfly species have average decline of 72%.
- Monitored moth species have average decline of 67%.
- Monitored vascular plant species have average decline of 28%, with one or two vascular plant species becoming extinct in any given County each year, particularly in the South and East of England.
- Water voles have disappeared from 94% of the places they have been previously recorded.
- There have been rapid losses of more than 50% in the last 25 years of once common species such as hedgehogs, house sparrows and common toads.
- Generalist species such as rats, able to thrive in a wide variety of environmental conditions and make use of a variety of different resources, are increasing in range and number, whereas specialist species, highly adapted to a narrow range of environmental conditions, such as some bumblebee species, are declining. Only six of 25 British bumblebee species remain widespread, with three now extinct.

The UK Environmental Change Network seeks to identify and understand long-term changes in UK ecosystems. Data were collected between 1993 and 2007 at twelve key sites ranging from upland areas such as the Cairngorms and Snowdonia to lowland sites located in southern England and Northern Ireland. Soils, vegetation and animal communities all showed indications of responses to environmental change over the study period:[64]

- All sites studied experienced increases in temperature over the analysis period.
- Butterfly species characteristic of warmer regions tended to increase at northern, upland sites, consistent with an effect of increasing temperatures.
- In contrast, ground beetles associated with cooler northern and upland areas showed declining populations.
- The acidity of rainfall was reduced, particularly at sites where atmospheric pollution is highest in the south of the country. This is characteristic of wider changes across the UK.
- Reductions in the acidity of rainfall were associated with a trend toward less acidic soils.
- There was no clear evidence of changes in plant communities in response to decreased soil acidity.
- Wetter weather in more recent years may explain a decline in short-lived 'weedy' plants at lowland sites, reversing an increase associated with drought in the early years of monitoring.
- Trends in nitrogen deposition differed between sites, but levels of ammonia (a nitrogen-containing gas released from intensive agriculture) remain high at some sites.

Although the effects of climate change on biodiversity are already evident in the UK and are expected to increase, there are uncertainties about precisely how, and by how much climate will change in different localities (POSTnote 343). Effective planning action would need to accept uncertainty and address the full range of variation in projected changes and their impacts. Long-term studies and monitoring of species and habitats are essential to improve knowledge of the impact of climate change, the more complex responses to adaptation measures and to inform decisions of policy makers. It is important that adaptation measures to conserve biodiversity in a changing climate are also integrated with adaptation measures that provide wider social and economic benefits (Chapter 5).

Conservation Gains in the UK

Despite declines for many habitats and species, there have also been success stories through targeted conservation action, such as the BAPs, particularly for some rare species, such as marsh harriers and cirl buntings. [65] The management actions undertaken to conserve particular species, such as the creation of ponds for amphibians, often result in wider biodiversity benefits.[66] Some species are also increasing in numbers due to environmental change, such as generalist species and some species that are spreading further north in response to climate change. This includes species (terrestrial and marine) migrating from the near continent into the UK (POSTnote 343).

There has also been a marked improvement in the management and condition of habitats protected by statutory designations in the last decade, particularly Sites of Special Scientific Interest (SSSIs). New habitat has also been created, such as various types of woodland, heathland, inland wetlands and coastal marshes. For example, since the launch of the UK BAP in 1995, over 800ha of reedbeds and over 3500 ha of species-rich grassland have been created, and more than 200ha of lowland raised bog has restored.[57] However, some types of habitat, such as ancient woodland, can only be re-created on timescales of hundreds to thousands of years, and in general, re-created habitat will support a lower level of ecosystem services than natural habitats (Section 5.5).

2.6 Ecosystem Thresholds and Biodiversity Loss

Loss of biodiversity could increase the vulnerability of ecosystems to other pressures such as climate change or ocean acidification (POSTnote 343). The output of benefits from ecosystem services is dependent on biodiversity, but the information on the specific aspects of biodiversity that must be maintained to ensure delivery of benefits is limited. There is a body of evidence to suggest that drivers of environmental change in natural resource systems, such as the agricultural modification of the quantity and quality of hydrological flows (Annex A), can increase the risk of exceeding ecological thresholds.[67]

However, the complex interactions of the multiple drivers of environmental change may limit understanding of how ecosystem processes and organisms respond to environmental change.[68] There is a lack of experimental evidence of the number of species performing similar functions required to act as ‘insurance’ in any given ecosystem, and the effects of novel interactions among species, such as the introduction of a pathogen or disease. Better understanding of how interactions between parts of food chains involve feedbacks may allow the processes that influence ecosystem service delivery to be better characterised (Annex A).

Despite these uncertainties, the risk remains that if biodiversity loss causes an ecosystem to become degraded beyond a threshold, ecosystem service provision will decline significantly, with impacts on human wellbeing. The potential for the delivery of a given ecosystem service benefit depends on certain ecosystem states and it is these that alter in response to drivers of environmental change, affecting the capacity of the ecosystem to sustain the delivery of ecosystem service benefits.

Linear and Non-linear Changes

An environmental limit is not necessarily associated with an irreversible threshold or regime shift. It can be applied to more gradual ecosystem degradation, where the levels of an ecosystem service benefit decline in proportion to the pressure applied to an ecosystem. With such linear relationships, it is relatively straightforward to monitor and set precautionary environmental limits as points along a continuous gradient of change, where there is sufficient scientific certainty and the impacts are usually reversible. Such limits can often be set in relation to the cost of reversing the damage. For example, it can be argued that the Somerset Levels have exceeded environmental limits for nitrification, given the cost of remediating the soils to pre-1930s conditions.

By contrast, abrupt non-linear thresholds can result in major transitions in ecosystems that may be extremely difficult to reverse or permanent. For example, thresholds for coral reefs in response to a number of pressures including sewage pollution and temperature changes are well known,[69] with periods of unusual warmth causing ‘coral mass bleaching events’ since the 1980s. At levels of CO_2 and acidification of seawater predicted for the coming decades, although tropical seawater will not become completely corrosive towards live corals, the ability of coral ecosystems to resist pressures will decline, as the balance between accretion and erosion is disturbed. The risk of a difficult to reverse and catastrophic loss of coral reefs will increase, requiring careful management of pressures placed on these ecosystems (POSTnote 343).

Abrupt shifts in ecosystem states often result from a combination of gradual changes in drivers, such as land use change, that appear to have little or no apparent impact up to a certain point, until an external shock such as storm, fire or disease outbreak, causes a threshold to be crossed. Non-linear thresholds can also be crossed gradually without a shock, but in both cases, the system becomes altered to the extent that it shifts from one set of mutually reinforcing ecological processes to another.

In Lake Veluwe, in the Netherlands in the late 1960s and early 1970s, the ecological condition of the lake hardly changed in response to increased nutrient levels until a threshold was reached. Beyond this point the water plants that had dominated the lake (charophytes) died off and turbid water conditions became established (eutrophication). The ecological condition was not restored until the nutrient levels were reduced far below the threshold level that triggered the regime shift. Eutrophication in freshwaters can be irreversible, or reversible only after massive reductions of phosphorus inputs for decades or longer, owing to the internal cycling of phosphorus within the lake system and the its accumulation in the soils of the catchment area.[70]

Impact of Abrupt Non-Linear Regime Changes
Such shifts pose a substantial challenge to the management of natural resource systems. They involve either changes in the ecological structures or in the way ecological processes occur. Simplified ecosystems that depend on a few or single species for critical ecological processes, such as landscapes dominated by human uses, appear more vulnerable to non-linear threshold changes.[67]

The number of studies that describe the impact of non-linear threshold changes on ecosystem services is limited, but they suggest that ecosystem services delivery may deteriorate. For most ecosystems, the relationship between interactions between organisms, different ecological processes and the output of different ecosystem services needs to be characterised better to determine at what point environmental changes will cause the delivery of services to fall below acceptable levels. [71]

Thresholds have been detected in oceans, freshwaters, forests, woodlands, drylands, rangelands and agro-ecosystems (Section 6.3).[72] Rangeland systems refers to expansive, mostly unimproved lands on which a significant proportion of the natural vegetation is grasses and shrubs, such as savannahs, where inappropriate grazing and fire management practices have been found to trigger the shift to scrub vegetation (Box 5).

Predicting Non-linear Thresholds
Unintentional ecosystem regime shifts are regarded as having significant impacts on human wellbeing. For example, the collapse of Canada's Newfoundland cod fishery in the early 1990s directly affected the livelihoods of some 35,000 fishers and fish-plant workers, led to a decline of over $200 million dollars a year in revenue from cod landings and had significant impacts on the local economy and society.

Thresholds are difficult to predict, due to limited understanding of how ecosystem processes and organisms respond to environmental change.[68] Modelling the system is difficult as ecosystems may show little change before the threshold is exceeded. For many ecosystems, the only way thresholds will be identified is when they have been crossed and a regime shift to an alternative ecosystem state occurs. Despite the varying degrees of uncertainty in where thresholds lie, it is possible to manage the risks of shifts by reducing pressures and enhancing ecosystem resilience, for example, by enhancing and maintaining biodiversity.

As with non-linear thresholds in other complex systems, such as financial systems, ecological thresholds are difficult to predict and avoid, as the state of the system may show little change before the tipping point is reached. Models of complex systems are not usually able to predict accurately where critical thresholds may occur.[73] For many ecosystems the only way thresholds will be identified is when they have been crossed and a regime shift to an alternative ecosystem state

Box 5 Managing Savannah Ecosystems

It has been suggested that landscapes can be scored on a scale for resilience to environmental change (see Chapter 5) ranging from "brittle" to "non-brittle", the scale reflecting annual humidity distribution, rather than the amount of rainfall, with brittle landscapes having erratic distribution of moisture throughout the year, even if some have high rainfall overall. Forty percent of the earth's land surface is covered by 'brittle' savannah grasslands, found in climatic regions of hot rainy summers and mild dry winters. They support a diverse assemblage of long and short lived grass species and a few woody plants. Brittle landscapes like savannahs, do not recover from disturbance with rest (the removal of the driver impacting the ecosystem), and the grassland is replaced by either woody vegetation or bare soil covered with algae and lichens, depending on the amount of rainfall.[74] A combination of brittleness and high grazing pressure can substantially increase the risk of desertification and erosion of soils.

The grass species in these ecosystems have co-evolved with large herds of herbivores, predated by packs of hunting animals, and need to be grazed to avoid shading themselves out and stifling seedling recruitment. The herds of grazing animals also break up the soil surface which would otherwise be baked hard, allowing seeds into the soils. In arid areas, seeds must be planted deeply or seedlings will die before their roots reach reliable water. Regime shifts between grass and shrub domination can occur if the existing competition between grasses and shrubs for water in the root zone is destabilised. These ecosystems need to be managed to maintain the grazing intensity at levels that would be expected in natural systems (neither under nor overgrazed). This will ensure a level of impact on soils and grasses that allows regeneration and that the less drought sensitive species are not removed, otherwise the proportion of bare soil increases. Appropriate grazing pressure, seed planting and nutrient recycling by herbivores are critical to maintaining sufficient root growth in the grass species and maintaining levels of carbon storage and other environmental functions such as water infiltration and flood protection. In addition to the grazing regime, the thresholds in arid to semi-arid savannah systems can be breached by differing patterns and intensity of fires and changes in drought occurrence affecting the competition between shrubs and grasses. [75]

occurs.[76] Some early warning signals are known for a number of ecosystem thresholds, such as the pattern of vegetation patches dying off near a threshold for catastrophic desertification in drylands.[77]

2.7 Global Environmental Limits

There is scientific evidence that total usage of natural capital by humans now exceeds the capacity of renewable natural resource systems.[78] Precautionary environmental limits could be set to avoid the degradation of ecosystems to levels that threaten human well-being. Such limits would be based on indicators that show the capacity or potential for delivery of particular ecosystem service benefits (Chapter 3, Annex B) and expert judgement as to the point or range of conditions beyond which the benefits derived from a natural resource system will be unacceptable or insufficient.[1]

Some researchers have called for environmental limits associated with the planet's biophysical systems to be defined to avoid threshold levels being breached and systems shifting into a new state (Table 2 and Figure 4).[79] For example, if the monsoon system were to shift as a result of climate change tipping points being breached, it could have deleterious consequences for human well-being.

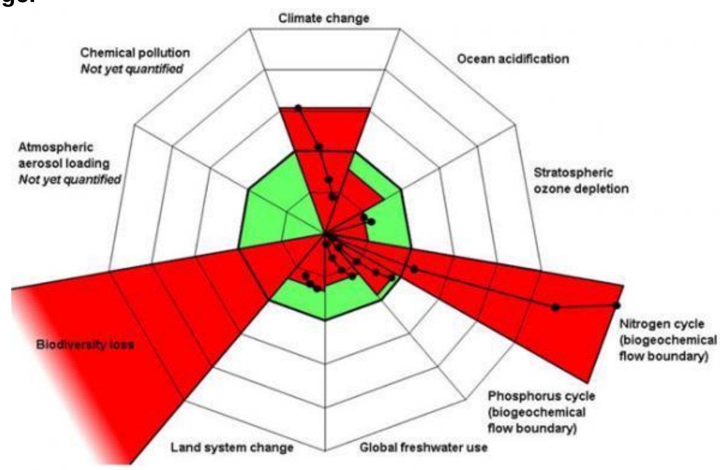
These limits are based on expert judgement and consensus on the point at which significant environmental risks are posed to human wellbeing (Chapter 6).The key planetary boundaries identified include:

- climate change
- rate of biodiversity loss (terrestrial and marine)
- interference with the nitrogen and phosphorus cycles
- stratospheric ozone depletion
- ocean acidification
- global fresh water use
- change in land use
- chemical pollution
- atmospheric aerosol loading.

Significant uncertainties surround where to set environmental limits, but it is likely that acceptable limits for climate change, rate of biodiversity loss and the nitrogen cycle have already been breached. Aggregate indicators for these categories could be included in national environmental accounting matrices to indicate the impact of national economies on global environmental limits, although this poses difficulties in measuring impacts embedded in products imported for consumption by any individual economy (Chapter 3).

In an increasingly resource-constrained world, exploitation of natural resource systems to yield one particular benefit for human wellbeing will be at the expense of other benefits provided by these systems, to the extent that ecological thresholds are exceeded. This places a greater burden on policymakers to consider the long-term implications of any given decision in terms of benefits gained and whether these outweigh likely losses and use of environmental limits provides one means of framing such decisions. However, there are considerable constraints to reducing rates of natural capital depletion, not least that the short-term consumption of natural resources to deliver economic growth remains a higher societal and government policy priority than the long-term maintenance of natural capital (Chapter 3).

Figure 4 Showing the Proposed Environmental Limits for Nine Different Categories of Drivers of Environmental Change.[79]



The green shaded polygon represents the safe operating space. Human activities have already pushed the Earth system beyond three of the suggested planetary environmental limits.

Table 2: Planetary Environmental Limits[79]

Biophysical System	Parameters	Proposed Boundary	Current Status	Pre-industrial Value
Climate Change	Atmospheric carbon dioxide concentration (parts per million by volume) Change in radiative forcing (watts/m^2)	350 1	387 1.5	280 0
Rate of Biodiversity	Extinction rate (number of species per million species per year)	10	>100	0.1-1
Nitrogen Cycle (part of a boundary with the phosphorus cycle)	Amount of nitrogen removed from the atmosphere for human use (millions of tonnes per year)	35	121	0
Phosphorus Cycle (part of a boundary with the nitrogen cycle)	Quantity of phosphorus flowing into the oceans (millions of tonnes per year)	11	8.5-9.5	-1
Stratospheric Ozone Depletion	Concentration of ozone (Dobson unit)	276	283	290
Ocean Acidification	Global mean saturation state of aragonite in surface sea water	2.75	2.9	3.44
Global Freshwater Use	Consumption of freshwater by humans (km^3 per year)	4000	2600	415
Change in Land Use	Percentage of global land cover converted to cropland	15	11.7	low
Atmospheric Aerosol Loading	Overall particulate concentration in the atmosphere, on a regional basis	Not yet determined		
Chemical Pollution	For example, amount emitted or concentration of persistent organic pollutants, plastics, endocrine disrupters, heavy metals and nuclear waste in global environment, or the effects on ecosystem and functioning of the Earth system	Not yet determined		

3 Accounting for Limits

Overview

- Natural capital constitutes environmental assets, such as forests, soils or marine habitats, from which beneficial services flow, supplying resources to the economy, such as agricultural crops and disposing of its wastes, such as treated sewage effluent. A minimum level of environmental assets is required to maintain the capacity of ecosystems to sustain ecosystem services important to human wellbeing at acceptable levels.
- If governments do not effectively monitor the use and levels of natural capital in national accounts ('environmental accounting'), the risks and probability of incurring costs through impacts on future economic productivity are not taken into account, nor are impacts on human well-being. Environmental accounting is one way of monitoring against environmental limits, but could also inform their development.
- However, appropriate methodologies are not yet fully developed nor the principles applied to decision making. In particular, better understanding of the mechanisms that link ecological systems to human well-being is required to inform integrated measurement and accounting tools, which can determine both the contribution of ecosystem services to national incomes and the expenditure required to maintain ecosystem service outputs.
- Procedures could be adopted to avoid the degradation of the natural capital assets to the level at which unacceptable risks to human wellbeing are posed. Alternatively, if the gap between the level of physical investment in natural capital required to maintain services and the level actually achieved could be determined, accounting methods could be used to calculate how far economies are from being within environmental limits.

3.1 Natural Capital

The term 'capital' is used to describe a stock or resource from which revenue or yield can be extracted. Natural capital constitutes environmental assets, such as forests, soils or marine habitats, from which beneficial services flow, supplying resources to the economy such as timber, agricultural crops or fish and disposal of its wastes, such as treated sewage effluent or carbon dioxide. Human wellbeing arises from a combination of types of capital: social capital, human capital and built capital; but these are all based on natural capital.

Natural capital can be degraded beyond critical thresholds (Section 2.5). It cannot always be restored or increased if degradation of ecosystems leads to irreversible changes, or species which have important roles in ecological processes become extinct. As well as supporting economic activity, natural capital, such as biodiversity, underpins ecosystem services and is critical to human wellbeing. When natural capital assets are depleted in quantity or degraded in quality, the flows of beneficial services to people are affected, for example, decreased catches from depleted fish stocks or decreased crop yields from degraded soils.

Critical Natural Capital

Four basic categories of natural capital are generally recognised: air, water (fresh, groundwater and marine), land (including soil, space and landscape) and habitats (including the ecosystems, flora and fauna which they both comprise and support).[80] The term 'critical natural capital' refers to the level of natural resources required to maintain the capacity of ecosystems to carry out processes and ecosystem services important to human wellbeing at acceptable levels.

Key processes include those of the production of biomass and oxygen, and the regulation of hydrological and atmospheric cycles. The quantity and the quality of natural capital affects the quantity and quality of ecosystem goods and services generated from these processes. For example, the recreational, amenity and other services delivered to the population around a 1km stretch of a river would be reduced if the quantity of water diminished significantly and/or if water quality deteriorated.

The use and flow of benefits from natural capital also vary spatially, so it also matters where natural capital assets are maintained or protected. For example, wetlands can absorb and slow flood pulses within a river catchment as well as maintaining river flows during periods of low rainfall (POSTnote 320). As with all natural capital stocks, wetlands provide benefits for the largest numbers of humans if located in areas from which these benefits can reach areas of high population density. If the wetlands were drained, these benefits would need to be supplied by the building of dams, levees and reservoirs, although each of these engineered solutions would have significant negative impacts on other ecosystem services.

3.2 Accounting for Natural Capital Depletion

Most commentators agree that the current systems of national accounting for the degradation of natural resource systems substantially underestimate damage incurred and overestimate savings from environmental protection (Annex B). For example, the economic activities that have the most significant effect on natural capital (such as energy generation, agriculture, fisheries, mining and transport) are subsidised by governments to an estimated one trillion dollars annually,[81] but the costs of the damage incurred as a result of these subsidies is not reflected in national accounts.

The Globe International Commission on Land Use Change and Ecosystems recently produced a Natural Capital Action Plan, for the 'Nagoya Declaration on Parliamentarians and Biodiversity' at the tenth Conference of the Parties to the CBD.[82] This recommended that a ministerial position should be created within finance ministries or treasury departments for managing natural capital. In addition, finance ministries or treasury departments should develop a comprehensive set of Natural Capital Accounts (NCAs) accompanied by a report that outlines which policy choices would be affected by integrating the true value of ecosystem services.

It also recommended an inter-departmental Ministerial Committee on Natural Capital to oversee (NCAs), advised by an expert technical advisory group. Individual government departments should be tasked with developing natural capital inventories of natural capital assets for which that department is responsible, with external auditors of government expenditure, such as the National Audit Office (NAO) in the UK, to issue public reports on the economy, efficiency and effectiveness of government policies concerning natural capital issues.[83]

To inform environmental accounts it needs to be established how natural capital and economy interactions manifest themselves in physical terms and how to select the appropriate data to describe these manifestations (Annex B). Valuation frameworks have been suggested as the basis for describing the stock and flow of natural capital for accounting purposes. However, in many frameworks, valuations are attached to the benefits arising from the flow of natural capital rather than the stock of natural capital (Section 3.3)

Any accounting framework needs to describe the stock *and* flow of natural capital to allow accounting and analysis of the interactions between the economy and the environment. With some forms of natural capital, such as forestry, the flow of benefits (timber) needs to be exploited at a rate which the overall stock (the forest) is maintained over time to avoid damaging the ecological infrastructure that supports it. The present value of a stock of natural capital is incorporates a measure of the future flows of benefits that it can generate.

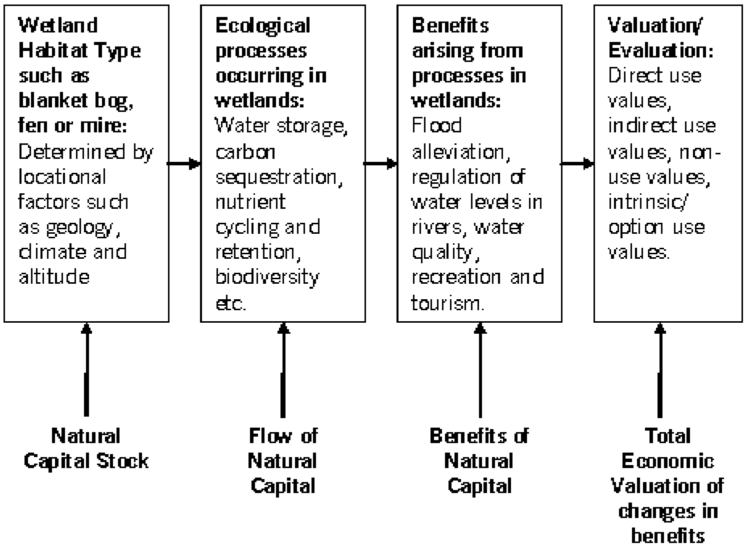
Accounts could describe changes in ecosystem quantity and quality either in physical units based on different indicators of ecosystem functioning (Annex B) or on changes in the monetary value of benefits flowing from that ecosystem. Over time, the stock or quantity of an ecosystem will change depending on the balance between human uses that transform or restore it, and the quality of the stock of ecosystems may change with the level of pressures impacting ecological processes. This could inform how national environmental policies and legislation could avoid breaching environmental limits through discouraging damaging economic activities and impacts.

3.3 Valuing Natural Capital

Natural capital assets perform a variety of basic supporting and regulatory ecological processes that contribute to human wellbeing. For example, water purification by a wetland contributes to provision of fresh water and food (through purification of water for crop irrigation and by sustaining freshwater fisheries) and to the tourism value of clean water courses and bodies.

A minimum level of wetland structure and processes is required, in terms of both area and quality, to ensure that such benefits are maintained at required levels (critical natural capital). Natural capital infrastructure, such as wetlands, therefore has economic value arising from its existence, as it is essential that such habitats are maintained to continue to provide beneficial services (figure 5). Some of the flows of benefits from the use of

Figure 5 The Stock of Wetland Natural Capital Maintains a Flow of Services Beneficial to Human Wellbeing.[84]



natural resources, such as agricultural commodities, are traded on markets. However, there are difficulties in obtaining widely agreed economic values for the benefits that natural capital provides when they do not have a directly traded market value, or where the traded value 'externalises' many hidden costs such as impacts on water resources and biodiversity arising from human uses of natural capital stocks.

Given their lack of market value, unregulated markets will undersupply activities that support natural capital assets and oversupply activities that reduce the stock of natural capital through environmental degradation, risking that biophysical thresholds will be breached at some future point. This market failure can be addressed through governance mechanisms that maintain natural capital stocks within agreed environmental limits.

The objective of placing economic valuations on ecosystems and natural capital stock is to inform decision making processes and to address this 'market failure' that occurs because of the public good characteristics of natural capital. If the valuation of natural capital does not reflect the serious and long term consequences of exceeding environmental limits, advice to decision makers, will not reflect the implications and trade-offs inherent in policy choices.[17]

Ecosystem Service Valuation

The purpose of ecosystem service valuation is still widely misunderstood. It is not intended to be a complete valuation of every aspect of the environment but to clarify the complex nature of interactions between humans and the environment. Valuation is not intended to displace the broader factors already present in environmental decision making frameworks, and most commentators agree that valuation of ecosystem services should be regarded as an additional component in decision-making rather than the sole mechanism.

Development of the concept of ecosystem services was intended to clarify why conserving biodiversity is important for human well-being. However, there is scientific uncertainty about how ecosystem interactions and services should be categorised and defined, as reflected by the lack of agreement on definitions of ecosystem services.[85] The term ecosystem service is applied by the Millennium Ecosystem Assessment both to outcomes that can be directly used and measured, such as water quality, and to processes that lack a distinct output in terms of human well-being, such as nutrient cycling.

Although it is broadly understood that ecosystem services are the contributions that ecosystems make to human well-being, there is a need to differentiate benefits, ecosystem services, ecological functions, and ecological structures and processes to characterise the mechanisms that underpin the links between natural capital and human well-being.

Structures, Processes, Functions and Services

Ecosystems are composed of physical, biological and chemical components, such as soils, water, organisms and nutrients (Box 1). Interactions among and within these components determine the capacity of a natural resource system to provide ecosystem services. Interactions between structures and processes, which may be physical (such as infiltration of water into soil), chemical (such as oxidation) or biological (such as photosynthesis), all involve biodiversity, although this relationship is not always straightforward (Figure 3, Section 2.4). These interactions give rise to ecosystem functions, intrinsic characteristics of the ecosystem, such as nutrient cycling, which are fundamental to its maintaining its integrity.

Current economic classification approaches are based on the assumption that services arise at the point at which an ecosystem provides an asset that is used by one or more human, with outputs such as nutrient cycling classed as ecosystem functions.[86] Such classification approaches avoid mixing the ecosystem functions and the services they give rise to, the means and ends, so there is no double-counting in economic valuations of natural capital stocks (Figure 6). They separate ecosystem interactions into two categories:

- those core ecosystem interactions that underpin basic supporting functions, such as nutrient cycling and retention, also referred to as "intermediate services"; and,
- interactions that underpin processes that directly benefit human well being, such as water quality, referred to as 'final services' or ecosystem services.

However, the ecological functions that contribute to an ecosystem service may also be a service in their own right, for example water quality is an intermediate service for other ecosystem services such as the provisioning service of fish for angling.

Valuation Frameworks

The TEEB framework separates the core ecosystem processes integral to ecological infrastructure and the beneficial processes directly linked to ecosystem service provision. In this framework, ecosystem services are defined as the aspects of ecosystems used (actively or passively) to produce human wellbeing, with the ecosystem service being the link between ecosystems and the things that benefit humans (the beneficial processes).[87] As human well-being may be an aggregation of different kinds of benefit, the TEEB valuation frameworks separates ecosystem services from benefits to clarify the role of ecosystems.

Ecosystem services are regarded as fundamentally ecological in character, with services transformed by other forms of capital, such as built capital, to realise benefits such as water treatment and distribution infrastructure for drinking water from clean water provision (Figure 6).[88] Several valuation frameworks similar to TEEB have been proposed by the US National Research Council, the Natural Capital Project, the US Environmental Protection Agency Science Advisory Board, the 'Valuing the Arc' project,[89] the French Council for Strategic Analysis and the and the UK National Ecosystem Assessment (Sections 4.4 and 4.7).

An alternative approach is to take account of different elements that vary according to context and ecosystem service to ensure consistent valuation and to avoid double counting, by defining structures, functions and processes needed to produce an ecosystem service (Section 4.5). The fundamental task is to understand the mechanisms that link ecological systems to human well-being (Figure 7). Most ecosystem services depend on a number of ecosystem functions and many more structural components and processes. They also have different spatial characteristics. For example, carbon sequestration is global in nature, whereas others, like pollination (box 2), depend on proximity, and can be used at source, or 'flow' to a different point of use.[90]

Society will also value ecosystem outputs differently in different places at different times. For example, agricultural ecosystems provide biodiversity, carbon storage, food provisioning, cultural and recreational benefits which will be differently valued by different groups, and these values will also differ over times in response to other factors. They also differ geographically, with the cultural importance of the outcome of particular agricultural practices, such as upland hay meadows in the Northern Pennines being valued more highly than food production. [91]

The capacity to deliver an ecosystem service exists independently of whether it is needed and only becomes a service when used by a set of beneficiaries. In defining what the 'significant' functions of an ecosystem are and what constitutes an ecosystem service, an understanding of spatial context and societal values is as important as understanding the structure and dynamics of ecological processes.[85]Although there is a growing literature on the valuation of ecosystem services, their scale and relative importance to society are still subject to uncertainty at the local and global scales and there is a lack of integrated measurement and accounting tools to determine the contribution of ecosystem services to national incomes. Broadly speaking, accounting frameworks require at least three things: the definition and measurement of quantities; the aggregation or adding up of those quantities; and weights for the individual elements in the aggregation index.

Figure 6 Classification of ecosystem processes and benefits[92]

With regard to ecosystem services, this entails:[93]

- defining the size of the system, including which natural capital stocks and their quantities are relevant to any given ecosystem service;
- the appropriate means of weighting these stocks to allow an accounting price to be estimated for each of them; and,
- the means of estimating the growth (or decline) in stocks and the conditions for the flows of ecosystem services from these stocks (on the basis of modelling).

Linking Ecological Systems and Human Well-being

Ecosystem services are a conceptual device that is helpful in understanding the transformations that link humans to natural capital by allowing a distinction to be made between the ecological functions that give rise to some benefit and the particular aspect of human well-being that is being considered. It is unlikely that any single approach can capture the complexity of ecological interactions to rank with certainty which of them are the most socially valuable. Further work is required to clarify the assumptions underlying valuation classification systems and to develop consistent definitions and a universally accepted ecosystem service typology, which clarifies the extent to which they are fundamentally dependent on biodiversity (Section 2.4).[91]

The difficulties in valuing the benefits of natural processes that have no market value and a historical legacy of a rights-based approach to environmental management, have previously favoured regulatory and legislative approaches to conserving natural capital stock, such as area-based conservation designations. This lack of valuation has led to a failure to consider the full impacts of the loss of natural capital in decision making, including the costs of the loss of benefits and the increase in environmental risks.

Policy decisions lead to actions that impact on natural capital via changes in ecosystem structure and function, which in turn alter ecosystem services. Policies to ensure more efficient natural resource use allocation decisions would be more effective if informed by the quantification of ecosystem services, taking account of uncertainty about future use. As stated in Chapter 2, there has been extensive degradation of natural resource systems over the last century and an understanding of the value of the benefits that have been lost will help to inform how these ecosystems should be managed to maintain future human well-being.

The classification and valuation of services could support the development of natural resource governance mechanisms and effective participatory decision making processes. For example, valuation can form part of negotiations around changes in ecosystem service provision (Section 4.10). Where a specific issue needs to be addressed, some classification of ecological processes for valuation purposes may inform policy decisions through cost benefit analysis, clarifying trade-offs for different stakeholder groups and the development of alternative scenarios arising from decisions on natural resource use.

Ecosystem Accounting

Incomplete understanding of how the interactions of ecosystem processes, structures and properties give rise to ecosystem goods and services remains a limiting factor in developing accounting frameworks. This is particularly the case for supporting services that do not give rise directly to benefits that can be easily valued, such as soil formation (Box 6). The development of national-scale accounting assessment for natural capital stocks and flows will need to be congruent with national income accounting the principles of the underlying ecology and consistently measured over time (Annex B). This would allow the effective procurement of environmental quality by governments and clear national measures of well-being arising from environmental public goods and market goods.[94]

The recent TEEB report has extensively discussed the options for developing indicators for natural capital and ecosystems within national accounting frameworks.[95] This builds on studies undertaken by the European Environment Agency (EEA), which has been developing and testing a system of ecosystem accounts as part of the revision of the UN System of Integrated Economic Environmental Accounting (Box 7, Annex B) undertaken by the UN London Group.[96]

Ecosystem accounting is intended to measure the following key elements:

- Is the ecosystem asset being maintained over time through natural processes both in terms of quantity (the stock of ecosystems) and quality (the capacity of ecosystems to maintain benefit provision) at levels consistent with the current and future requirements of society?
- Is the full cost of maintaining the stock and quality of ecosystems covered by the current price of goods and services produced for the economy?

Box 6 Valuation of Soils

Soil is a complex, variable and living media that acts as interface between biogeochemical processes occurring in the atmosphere, geological bodies and water environments. Soils and sediments are key slow variables that regulate ecosystem processes by providing resources to other organisms, such as plants. It is argued that the non-marketed as well as marketed benefits of natural resources such as soils should be considered assets by decision-makers. Soils are important natural capital assets from which many benefits flow including fertility provision, pollution attenuation, carbon sequestration, water storage and flood regulation. Sustaining soils is fundamental to maintaining many ecosystem service benefits, so valuation must therefore encompass its impact on other resources and the broader interconnections with other natural capital benefit flows.[97]

Soil comprises mineral particles, organic matter, water, air and living organisms. The quantity of soil in an ecosystem depends largely on the balance between inputs from weathering (the breakdown of rocks to form soil) or deposition of sediment by floods and losses from erosion, the average rate of soil formation being less than 1cm depth per century. Organisms, especially plants, add organic matter to soils through decomposition of dead tissues. Natural imbalances between the outputs and inputs means there are deeper soils in floodplains and at the base of hills than on hilltops. However, in general, changes in natural soil capital are very slow where the natural vegetation cover is intact, maintaining organic inputs and limiting erosion outputs. The quality of soils, in terms of their physical and chemical properties, are also critical to ecological process. For example, organic matter and fine particles of mineral soil play roles in retaining nutrients and water, but are concentrated near the soil surface making them vulnerable to erosion. Freshwater and marine systems are heavily affected by soil erosion, with increased sediments in water courses leading to nutrient enrichment (eutrophication) and transfer of other pollutants such as pesticides.

Human changes in land cover increase erosion rates, which can lead to the rapid loss of soils within decades. Human activities, such as the modification of river channels to reduce flooding, may lead to reduced input of sediments into soils. Management of soils within urban areas often consists of concreting over, paving or compacting to seal them, resulting in the ecosystem service benefits being lost, including irreversible losses of soil fauna within human timeframes. Increased soil sealing is a key factor in the hydrology of urban areas, affecting surface water drainage and flood risk (POSTnote 289) and aquifer recharge, and reduces the availability of water for surface vegetation.[98] Studies suggest that current levels of soil degradation occur because markets fail to account for the social cost of poor soil management as well as the non-market benefits. For example, soil fauna, (such as earthworms), enhance soil drainage and create passages for roots, aerating the soil, and recycling organic matter and nutrients. However, as the economic benefits they provide are not taken account of by markets, private activities will continue to damage soil fauna.[99] Defra recently commissioned a report that attempted to value the benefits of soils, although there are substantial knowledge gaps about changes in soil processes and benefit provision, values could be derived for:[100]

- Carbon storage and sequestration benefits, with globally three times as much carbon stored in soils than is in the atmosphere. Increased temperatures due to climate change are likely to increase the rate at which organisms degrade organic matter in soils in temperate areas and may result in these soils moving from being net stores of carbon to net emitters. The UK's peatland soils are a particularly important store of carbon, with the eight percent of the UK covered with blanket peat moorland storing most of the UK's terrestrial carbon. Peat soils in England are estimated to store 296 million tonnes of carbon, roughly equivalent to two years of total UK carbon emissions.[101] In an undamaged state, peat remains wet at the surface all year, sequestering between 0.1 -0.5 tonnes of carbon per hectare per year.[102] However, peatland soils are being degraded by agricultural use in the lowlands, such as the Anglian Fens, and by drainage and burning in the uplands, and under these conditions become net emitters of large quantities of carbon.[103] The value of non-marketed benefits of carbon storage appears to exceed marketed benefits from the uses that degrade peatlands.
- Water storage and flow remediation benefits, which significantly reduce flood risk. The value of the soil in water and flow mediation is significantly changed by land uses that impede infiltration because of compaction or surface capping with impermeable substrates such as asphalt.
- Nutrient cycling and crop production benefits, such as the fixation of atmospheric nitrogen into organic nitrogen compounds by soil bacteria. These were found to have substantial value. The costs of deviation from best soil management practice with resultant soil degradation can be directly measured through lost yields for different crops.
- Supporting construction benefits, which vary with soil properties but have little impact on land values and the costs of development, given the high costs of development land in comparison to other land uses. Such development will substantially reduce the other benefits provided by soil.
- Natural attenuation of pollution and contamination benefits, which arise from soils as they absorb pollutants, such as pesticides, and degrade them to less toxic compounds. However, this is a complex process affected by many factors such as pollutant type, soil structure, levels of organic matter, clay content, microbial biomass, hydrology, changes in land use and climate. This creates difficulties in determining the level of these benefits provided by soils, but it is clear that poor soil management substantially reduces the level of these benefits from soils.
- Archaeological and landscape heritage protection benefits, which is reflected in the social value of soil preserving archaeological sites and artefacts as shown by the public willingness to pay for archaeological sites to be preserved through agri-environment schemes (Section 5.3).
- Support of ecological habitats and biodiversity benefits, which is reflected by the high social value placed on the landscape and habitat in UK farmland, estimated to be worth £845 million. The degree to which soil properties directly contribute to valued landscape and biodiversity is difficult to determine, but it is clear that poor soil management directly contributes to the degradation of habitats and loss of biodiversity.

- How is the flow of ecosystem goods and services supplied for human use factored into the overall calculation of wealth or social wellbeing.

To achieve ecosystem accounting there will need to be appropriate metrics to determine the amount and quality of ecosystem assets, the level of these assets required to meet society's requirements and suitable metrics for determining the gap between requirements and existing ecosystem assets (Section 3.5). Ecosystems can provide two types of output in the broadest sense: marketed ecosystem goods in the form of provisioning services such as food, and goods that provide public benefit but are un-marketed, for example, regulating services such as soil formation.

This latter category includes the processes required for ecosystems to maintain their basic integrity and continue to provide services. The EEA have proposed a framework of metrics required to ensure both that ecosystem integrity is being maintained and that the required output of ecosystem service benefits is being achieved (Annex B). This framework was used to show ecosystem accounting could be implemented in four southern Mediterranean wetland test studies.[104]

The aim of constructing the accounts was to allow the assessment of whether the value of natural capital represented by the wetlands was changing over time (Box 8), with the services associated with these ecosystems categorised in terms of the strength of their link to biodiversity. The accounts can also help to determine whether ecosystem service benefits are at levels acceptable to society or in breach of environmental limits. The environmental limit is represented by the minimum level of natural capital required to generate the level of ecosystem service benefit required by society.

Common International Classification for Ecosystem Services

The EEA has suggested a Common International Classification for Ecosystem Services (CICES) could be developed. This would be consistent with accepted typologies of ecosystem goods and services, such as the MA, and compatible with the design of integrated environmental and economic accounting methods being considered in the UN System of Environmental and Economic Accounts (SEEA, Box 7 and Annex B). CICES aims to describe the links between ecological structures and processes and the benefits that flow from them.

CICES is intended to describe the connections between the biological and physical components of ecosystems and the various products, activities that are wholly or partly dependent on them, illustrating the ‘pathway’ from ecosystems to human well-being while avoiding the issue of double counting (Figure 7). Since its aim is to identify the final products of ecosystems, only three broad thematic categories are suggested as the basis of CICES - provisioning, regulating and cultural outputs. However, these can be further subdivided into nine generic classes, such as nutrition or regulation of wastes, which can be cross referenced to existing standard classifications for activities and products used in the System of National Accounts.[105]

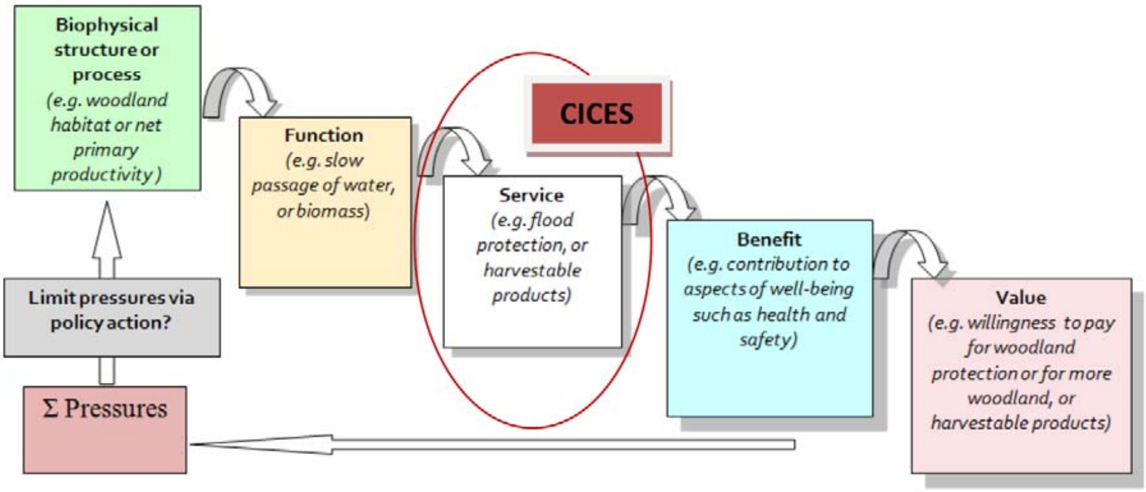
Box 7 The UN System of Environmental and Economic Accounts (SEEA)

The 2003 SEEA handbook provides a framework for incorporating the role of the environment and natural capital in the economy through a system of satellite accounts to the system of national income accounts (SNA). The SEEA has a structure similar to the SNA, covering both stocks and flows of environmental goods and services. Most environmental assets are not traded in markets, and those that are, such as mineral deposits, fish in the ocean or timber from tropical forests do not have their depletion factored into SNA asset accounts. The SEEA corrects this by including the depletion of traded natural resources and calculating the cost of depletion according to conventional economic rules. However, valuation remains imprecise for other forms of environmental degradation. The SEEA 2003 maintains four categories of accounts:

- natural resources asset accounts, which record the volume and economic value of these stocks, and changes in them, in both physical and monetary terms.
- flow accounts for environmental management and protection, which provide information at the industrial sector level about inputs such as the use of energy and materials into production (including the input of non-market environmental services) and outputs in terms of pollutants and solid waste.
- expenditure on environmental protection and natural resource management and other environmental financial transactions, including taxes, fees and property rights in relation to the environment.
- valuation of non-market flow and environmentally adjusted aggregates: present non-market valuation techniques and their applicability in answering specific policy questions, and discusses the calculation of several macro-economic aggregates adjusted for depletion and degradation costs.

Physical accounts help to set priorities for policy based on the volume of resource use, pollution and so on, while monetary accounts identify the relative costs and benefits of reducing pollution or resource use. Most developed countries focus their environmental accounting efforts on pollution damage and control costs and material and energy flows in their economies rather than on the depletion and degradation of natural resource systems. This is partly due the difficulties in pricing natural resources that have no market value, such as biodiversity or climate regulation, and the export of resource depletion impacts to other, usually developing, countries. Tracing the environmental impacts of products and commodities in the country of origin poses difficulties in obtaining actual data, and impacts often have to be modelled, incorporating a number of uncertainties. As such, the 2003 accounts do not recognise the existence of irreplaceable natural capital or the need to preserve ecosystems and their services. The SEEA is due to be revised in 2012, with the new version split into two volumes, the first a set of standardised methods for environmental accounting that can be integrated with the System National Accounts (SNA), which will include the four categories from the 2003 SEEA. The second volume will cover the areas where there is not yet a standardised methodology, but where numerous approaches have been developed, such as ecosystem valuation (Annex B).

Figure 7 Showing the Relationship between Ecosystem Functions, Services and Benefits, and the Context for CICES



The aim of the classification is to develop a flexible structure that broadly links to the categories of ecosystem service that are discussed in ongoing international initiatives such as TEEB. The need for CICES arises because at present there is no accepted definition of classification of ecosystem services for accounting frameworks. The current typologies of ecosystem goods and services are designed for making assessments and valuations at a project and policy level, rather than as a framework to link to the classification systems used in economic and environmental accounting.

Challenges to Developing Methodology

National accounts require the systematic description of both the costs and benefits associated with ecosystem service provision. To sustain the flow of an ecosystem service, there needs to be sufficient reinvestment in the stocks of natural capital, which represents the 'costs' of ecosystem service provision (Figure 8). For example, management measures to conserve biodiversity at levels sufficient to maintain the flow of an ecosystem service (Section 2.5) is a cost of provision of benefits from ecosystems.

If the scale and/or value of the intermediate services consumed in the production of final goods can be identified, the level of 'reinvestment' in natural capital needed to sustain the output of relevant ecosystem services could be determined.[106] The 'reinvestment' in natural capital may take a number of different forms other than the costs of management measurements, including protection, restoration, or forgoing the use of natural capital assets to ensure that natural resource systems retain their capacity to renew themselves.

Box 8 Ecosystem Accounting[104]

The European Environment Agency has carried out a methodological test study on Ecosystem Accounting for the Mediterranean Wetlands, which concluded that:

- Ecosystem accounts can be implemented across the three geographical scales most relevant to prevailing governance models and societal welfare considerations - the global/continental scale, the national/regional and the local scale.
- From a policy and statistical perspective, environmental accounting should be prioritised from a top down rather than a bottom-up perspective.
- Simplified, global-scale ecosystem accounts can be used to assess losses in physical units and the costs of restoring the ecosystem for maintaining their functions and their capacity for providing a flow of services in the future.
- Ecosystem accounts could be integrated with national economic-environmental accounting procedures.
- Ecosystem accounts would be helpful for planning departments and environmental protection agencies to internalise fully environmental considerations in decision making (through for example cost-benefit analysis, section 4.7).
- Land use systems can be used as an analytical unit for accounting, reflecting the interaction between ecosystems and humans (Section 5.4). Stocks and flows of land cover, water, biomass/carbon and species/biodiversity are key accounts for calculating the ecological potential of terrestrial social ecological systems (Section 5.2).
- Asset valuation is a useful means for judging trade-offs between ecosystem services in decision-making. However, the maintenance of ecosystem service levels does not necessarily require economic valuation of the service, merely physical measurement to observe any degradation in potential.
- Maintenance of ecosystem capital requires consideration of expenditure on environmental protection and resource management and the additional costs needed to mitigate potential ecosystem damage when this expenditure is not sufficient. These costs can be used as an alternative means of economic valuation.

The lack of such information about the level of unexploited natural capital required to maintain the capacity of ecosystem services at acceptable levels is a major impediment to the development of effective accounts. Only a few studies are currently available that look at the costs of ecosystem service maintenance.[71]

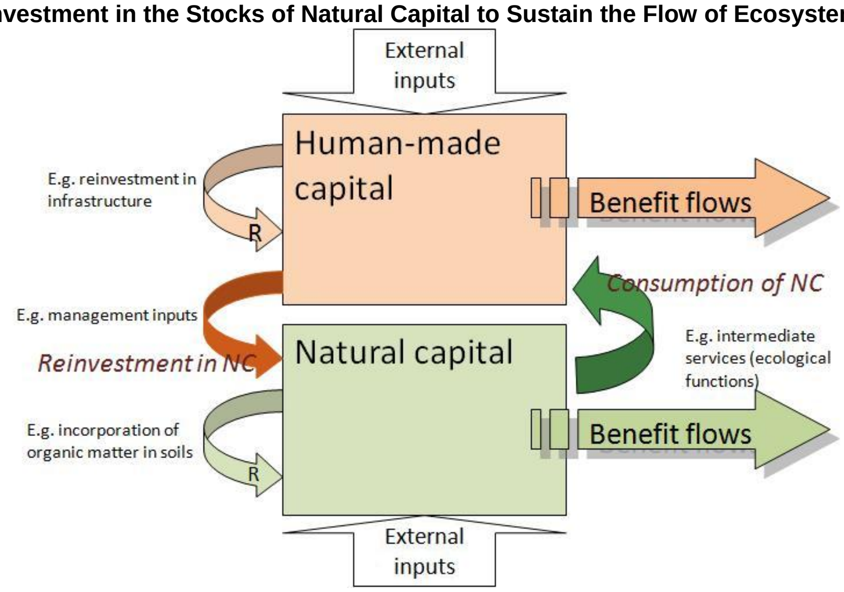
Specific challenges to developing methodologies include:

- disentangling the value of ecosystem services from the benefit which it delivers (for example, water quality and drinking water);
- most economic activities will only impact on ecosystems in a small way along with lots of other small impacts from other economic activities;
- valuing ecosystem services where financial payment systems are emerging (such as carbon trading) and valuing ecosystem services that have no market value (Section 4.7);
- the low incremental economic value of individual environmental impacts versus the long-term cumulative environmental cost of such impacts. For example, although a small area of forest may have a low economic value, the cumulative impacts of the losses of lots of small areas of forest over an extended period will have large impact on the environmental functions supported by regional forests as a whole; and,
- how changes in natural capital area or stock affect different ecosystem service outputs. For example, the loss of 50% of a forest would halve the provisioning service of wood production, but could lead to far greater loss of other services, such as recreational services.

Ecosystem assessments (Section 4.4) may be able to provide a physical assessment of the quantity and quality of stocks of natural capital for accounting frameworks. However, the complete monetary valuation of natural capital would require the disentangling of the valuations for environmental functions and the ecosystem services they give rise to, as well as the market values of the commodities arising from ecosystem assets, as is being attempted with CICES.

It is not possible at present to implement accounting methods for every ecosystem service benefit. This is likely to require considerable greater understanding of ecological processes and how to value them than is currently available. Greater understanding of ecological processes is also required to identify ecosystem service indicators that reflect how they are impacted by different types of external pressure (Annex B). The system will have to be developed on a step-by-step basis as understanding of individual ecosystems increases, rather than starting from a standardised model.[107] With flexible accounting frameworks, such as CICES, it is possible to accommodate new understanding as it arises.

Figure 8 Reinvestment in the Stocks of Natural Capital to Sustain the Flow of Ecosystem Services.[108]



A flow of natural capital (such as timber) is required for the production of human made capital (such as buildings) and maintaining natural capital stock (such as forests) through protection and management is required to sustain this flow. The flow of services have to be combined with human capital, such as manufacture capital to produce a final benefit.

3.4 UK Environmental Accounts for Agriculture

The UK has been developing aggregate accounts of the impacts of agriculture on the environment for over a decade. Agriculture's role in the UK economy is small, accounting for only 0.8% of the Gross Domestic Product in 2000, not including the food processing and related sectors. 2008 was a particularly favourable trading year for agricultural commodities and the total value of UK farm production including non-food products was £19.8 billion. The use of natural capital by UK agriculture is substantial as it accounts for a high proportion of land use (about 74% of land in the UK and 70% in England). The UK has produced aggregate agricultural accounts since 1973, which have included measures of the impacts of agriculture on the environment since the 1990s (Box 8).

The environmental accounts for agriculture provide a framework for measuring and valuing its positive and negative impacts on the environment. Negative impacts on the environment include the emissions of pollutants and greenhouse gases; the abstraction and use of water; the degradation of soil structure and biodiversity; and generation of waste. Positive impacts include "agri-environment schemes" (Section 5.3) or traditional land management measurements that support ecosystem service benefits.

The intensification of agriculture has substantially improved productivity but has increased negative impacts. Further intensification of agriculture in response to global food and bioenergy markets is likely to be a key driver of ecosystem change in the near future. Between 2004 and 2007, the area of oilseed rape grown in England under the energy aid payment scheme for conversion to biodiesel increased from 10,862 to 240,032 ha (from 0.2 to 4.9% of total croppable area).

Baseline of No Agriculture

Farming activities carried out in an environmentally responsible manner can enhance and maintain the characteristics of landscapes, particularly in terms of farmland biodiversity, with much of UK flora and fauna having adapted to traditional agricultural practices. The valuation framework for the accounts assumes a baseline of no agriculture, translating as no agricultural activity and zero impacts such as water or air emissions as well as zero provision of benefits such as landscapes, habitats and species associated with agriculture.

This does not imply that a zero level of agricultural activity is desirable, but simply captures the positive and negative effects arising from the current level of agriculture. By assigning monetary values to these, they can be compared with each other and aggregated to give a measure of the net overall annual impact (Table 3), which provides an estimate of the total natural capital stock and flows affected by agricultural activities.

Box 9 UK Environmental Accounts for Agriculture

The accounts aggregate existing robust valuations for environmental costs incurred by agricultural impacts with physical indicators. Certain values were established through benefits transfer, which is the application of existing economic value estimates from literature and applying them in a similar but differing context. Some of the values are based on actual costs, for example, the costs incurred by other sectors in clearing up actual agricultural pollution incidents, such as the Environment Agency's spending for replenishing affected fish populations following pollution incidents. Some costs, such as those incurred by water companies to clean water polluted by agricultural activities are relatively straightforward to transfer. Other costs are estimated using data on individuals' (or households') Willingness to Pay (WTP) to maintain the current quality and quantity of environmental assets, to avoid a negative environmental impact or to secure a positive one. This can be calculated from market data for those environmental impacts that are traded or reflected in actual markets, such as the cost of water used by an individual/household. Alternatively, in the absence of market data, estimates can be derived from surrogate markets, such as the prices of properties overlooking a water body, the costs of travel for people to enjoy services, such as tourism and leisure opportunities, or from the direct surveying of attitudes using questionnaires. Surrogate values are not used in the UK environmental accounts, which incorporate only survey data.

In general, productivity gains or losses from agricultural activities accruing to other economic sectors are assessed through actual market values, whereas environmental or social gains and losses are based on survey values. However, calculating the environmental impacts from agriculture requires many assumptions. Estimates therefore have a high degree of uncertainty, and should be regarded as only indicative of the flows of natural capital stock. It is likely that the inclusion of new data and methodologies in revised accounts in coming years will lead to substantial alterations to the estimates made. Once further developed, the environmental agricultural accounts could provide a measure of the environmental sustainability of agriculture and its contribution towards meeting environmental targets.

One of the policy applications for the accounts could be in justifying payments to farmers to protect water catchments through relevant measures such as blocking of moor grips (Box 14) or reducing fertiliser application. The water quality costs in the 2000 to 2007 accounts are aggregated with the physical indicator of the General Quality Assessment (GQA) exercise. In line with the Water Framework Directive, water bodies are now assessed for "good ecological status" requiring the monitoring of additional biological, physical and chemical criteria, with costs of reaching this status rising consequently in future accounts. In PR09 (the water industry price review looking ahead to the 2010-2015 accounting period), OFWAT has permitted farmers to be paid to deliver specific environmental benefits, such as reducing water colouration and improving the condition of SSSIs in the case of the SCaMP1 project in the Forest of Bowland (Box 14).

Table 3[109]

Environmental Accounts for Agriculture - United Kingdom

Current price table

The current price table shows the estimated valuations in each year using that year's prices

£ million

		2000	2001	2002	2003	2004	2005	2006	2007
Benefits									
Landscape	*semi-natural habitats*	*463.1*	*472.8*	*486.5*	*489.8*	*506.8*	*526.7*	*562.5*	*577.5*
	linear features	*25.3*	*25.6*	*26.0*	*26.7*	*27.4*	*28.1*	*28.9*	*30.1*
	Total Landscape	488.3	498.5	512.5	516.5	534.2	554.8	591.4	607.5
Biodiversity	*Habitats (A/SSSIs)*	*247.4*	*275.3*	*303.5*	*336.4*	*385.4*	*417.8*	*461.0*	*504.6*
	Species	*535.4*	*542.3*	*548.3*	*565.0*	*572.2*	*583.5*	*595.2*	*586.3*
	Total Biodiversity	782.7	817.6	851.8	901.4	957.7	1001.3	1056.2	1090.8
Other benefits	*Waste sink*	*19.4*	*25.3*	*25.8*	*28.3*	*30.8*	*36.3*	*37.4*	*39.0*
	Total benefits	1290.4	1341.4	1390.1	1446.2	1522.7	1592.3	1685.1	1737.4
Damages/costs									
Annual flows									
Water quality	*Estuarine*	*3.6*	*3.4*	*3.4*	*3.3*	*3.2*	*3.2*	*3.3*	*3.5*
	Lake	*22.8*	*23.2*	*23.6*	*24.3*	*25.0*	*25.7*	*26.6*	*27.7*
	Marine	*2.4*	*2.5*	*2.1*	*0.9*	*1.3*	*1.0*	*0.4*	*2.3*
	River	*66.3*	*63.4*	*67.0*	*70.2*	*71.3*	*68.3*	*66.4*	*69.3*
Water pollution	*pollution incidents*	*1.1*	*0.8*	*0.6*	*0.8*	*0.7*	*0.4*	*0.4*	*0.4*
Water abstraction	*Abstraction*	*52.9*	*44.5*	*45.3*	*56.8*	*45.3*	*38.4*	*39.6*	*41.3*
Drinking water	*clean up costs*	*79.4*	*102.9*	*145.9*	*151.7*	*130.7*	*118.9*	*116.8*	*131.8*
Flooding	*flooding from agriculture*	*201.0*	*204.6*	*207.9*	*214.0*	*220.3*	*226.6*	*233.8*	*243.8*
	Total water	429.5	445.3	495.9	522.0	497.8	482.6	487.3	520.1
Other damages	*Waste*	*7.0*	*7.2*	*7.3*	*7.5*	*7.7*	*7.9*	*8.2*	*8.5*
	Soil erosion	*9.1*	*9.3*	*9.4*	*9.7*	*10.0*	*10.2*	*10.6*	*11.0*
	***Damages - annual flows**	**445.7**	**461.8**	**512.6**	**539.2**	**515.5**	**500.8**	**506.1**	**539.6**
Present values									
GHG	*Carbon dioxide*	*71.8*	*77.1*	*80.9*	*87.2*	*91.9*	*95.9*	*98.8*	*103.1*
	Methane	*307.2*	*305.9*	*320.6*	*349.0*	*376.9*	*397.4*	*433.8*	*452.4*
	Nitrous oxide	*461.8*	*456.6*	*490.4*	*520.9*	*559.3*	*599.7*	*611.6*	*637.9*
	Land use change	*0.1*	*0.1*	*0.1*	*0.2*	*0.2*	*0.2*	*0.2*	*0.2*
	Total GHG	840.9	839.7	892.0	957.2	1028.3	1093.2	1144.4	1193.5
Air quality	*Ammonia*	*432.8*	*435.1*	*436.1*	*435.2*	*459.4*	*459.2*	*477.1*	*497.5*
	other air emissions	*162.9*	*156.7*	*157.7*	*154.7*	*149.1*	*142.0*	*130.8*	*136.4*
	Total air quality	595.6	591.9	593.8	589.9	608.5	601.2	607.8	633.9
	***Damages - present values**	**1436.5**	**1431.5**	**1485.8**	**1547.1**	**1636.7**	**1694.4**	**1752.3**	**1827.4**

*Present values and annual flows are based on different methodologies and cannot be aggregated.

The impact categories include water, air, soil, landscape, habitat and species, and waste. The information gaps in the accounts mean that the valuations derived at present are only indicative of the costs of the impacts rather than a comprehensive assessment. Despite these shortcomings, the accounts could be used further to inform policy decisions by identifying negative trends to be addressed and the effectiveness of measures to do so.

The accounts suggest that the net environmental costs of agriculture in the UK have been decreasing in real terms since 2000. Annual environmental costs were about £2.57 billion in 2007, mainly associated with soil emissions to air and water related damages. Valuation could be used to show the impact of particular policy decisions associated with land use. For example, the costs of soil degradation are estimated at between £250 and £350 million/year, mainly associated with soil erosion, carbon loss and the costs of dredging rivers and water treatment.

Policy Uses of Environmental Accounts for Agriculture

For national accounting purposes, the no agricultural activity baseline is appropriate to elicit total benefits and costs and to reflect the full economic implications of the sector's impact on the environment. The accounts have also been used to contribute to policy development on the future of the CAP.

However, the counterfactual scenario, an estimate of what would have occurred in the absence of the activity being evaluated, used to baseline accounts is critical to what else the accounts can be used to inform. Other counterfactual choices are more appropriate for comparing differences in the costs and benefits of different agricultural policies, and to indicate the resources that are substitutable, which trade-offs between benefits would be acceptable and possible synergies in management objectives. Actions at a local scale can have wider impacts at a regional or national scale, so counterfactuals should also be scale-specific.

In the case of landscape and habitats, the counterfactual can be a wild unmanaged landscape or habitats, but arguably, landscapes or habitats managed in a specific way could yield greater benefits for human wellbeing than an unmanaged one and would be a more optimal counterfactual. Sector-orientated accounting should be able to indicate which agricultural activity is optimal in a given landscape area. With the appropriate baseline, the accounts have the potential to make explicit the public benefits delivered by agriculture, such as biodiversity and landscape, as a justification for continued subsidisation.

The eventual aim is to incorporate the environmental accounts for agriculture into national accounts. Before this can be achieved, information gaps need to be addressed to increase the robustness of these novel environmental accounts. There remain gaps in the data both in terms of physical indicators, such as for biodiversity, and of economic valuation studies to draw on, such as the economic value of the landscapes created by agriculture. Examples of landscapes include lowland cropped or fields grasslands. Defra is proposing to commission a study to fill the landscape gap, which will value six different types of landscapes throughout England. The main biodiversity indicator used in the accounts, the farmland bird index, does not cover all farmland species and the accounts require a better estimate of these.

3.5 Acceptable Limits of Benefits from Ecosystem Services

The world total of environmental depletion and degradation costs has been estimated to be about 6% of world GDP annually, or $3 trillion in 2006. Natural resource depletion accounts for 89% of this, with the remaining 11% being greenhouse gas emissions and other air pollutants.[27] The extent of natural capital loss has been summarised in the recent TEEB report, which estimated failure to halt biodiversity loss on land, which may cost $500 billion by 2010 (the estimated value of ecosystem services that would have been provided if biodiversity had been maintained at 2000 levels). At sea, unsustainable fishing was estimated to reduce potential fisheries output by an estimated $50 billion/year.[110]

Without changes in institutions and market incentives, further declines in natural capital are likely. Those nation states which gain from actions that deplete natural capital will continue to avoid paying the full costs of their actions, with the burden of the costs of depletion falling future generations as well as more severely on the least developed countries. An understanding of the minimum levels of ecological functioning required to maintain ecosystem service provision at acceptable levels, and the level of reinvestment in natural capital required to achieve this, provides a way in which the notion of ecological limits or thresholds could be incorporated into environmental accounting.

At a national level, defining the proportion of benefits from ecosystem services being provided at acceptable levels above limits for inclusion in an accounting system is likely to be challenging. In some cases, environmental limits can be defined from relevant scientific understanding of ecological thresholds. For example, when considering the recreation service from salmon populations, water quality thresholds exist that determine whether salmon populations can be present or not in a river.

In most cases though, current levels of understanding are insufficient to define the ecological thresholds at which levels of pressures on ecosystems will change them to a state that will not support existing levels of beneficial outputs. This suggests a precautionary approach is required to the protection of natural capital assets. However, it is also necessary to define social and economic limits through the participation of the stakeholder groups that stand to gain or lose from changes in levels of different benefits from ecosystem services.

The importance of the delivery of a given service for stakeholders is a means for setting precautionary limits where there is uncertainty as to where ecological thresholds lie for management purposes at a local or regional level (Chapter 4). However, the limits would be insufficiently robust and unwieldy for national accounting purposes. Any large set of summary indicators of ecosystem services (Annex B) would also need to be further combined into a simple index of ecosystem service provision for national accounting requirements, e.g. x% of ecosystem services are providing benefits at acceptable levels, to determine the status of natural capital at a national level.

The Sustainability Gap

Alternatively, if the gap between the level of physical investment in natural capital required to maintain services and the level actually achieved could be determined, accounting could be used to calculate how far economies are from being environmentally sustainable, usually referred to as the sustainability gap (SGAP).

This gap in expenditure would reflect the shortfall of reinvestment an economy needs to make to maintain, protect and restore ecosystem services to the level needed (within environmental limits). It could also be in the form of 'use forgone' of the stock of natural capital that must not be appropriated to ensure that ecosystems retain their capacity to renew and sustain themselves. These costs are different from expenditure on management or protection of a given ecosystem, as they represent the expenditure required to restore ecosystem service benefits to required levels, both within the UK and for any imports of services used.

The sustainability gap methodology has been demonstrated for various environmental impacts such as climate change, ozone depletion, acidification and waste disposal.[111] There are major challenges to identifying appropriate targets for ecosystems and environmental functions, in particular, determining the level of reinvestment in natural capital to sustain the output of ecosystem services.[112]

For the sustainability standards to be calculated, there would need to be robust indicators of ecosystem functioning (Annex B) that would accurately reflect the maintenance of key ecosystem services functions. In addition, scientific definitions would need to be agreed on what constitutes 'healthy functioning ecosystems' as required by Defra's action plan for securing a healthy natural environment.[113] Criteria that need to be considered when setting the limits include:

- critical ecosystems and biogeochemical systems not being threatened;
- no detrimental effect on human wellbeing;
- renewable resources not being harvested faster than they can regenerate; and,
- non-renewable resources not being depleted faster than they can be substituted.

From the difference between these limits and actual measures for the different environmental categories in national green accounts, as indicated by relevant indicators, the gap in sustainability could be measured. This is both in terms of how far it fell short of the standard and how many years it would take to reach this at current rates of abatement/expenditure. The limit would set the level of critical natural capital to be maintained and how much it would cost, as opposed to current policy approaches that attempt to balance environmental protection against economic growth. Individual policy decisions could be assessed on the basis of whether they take one nearer or further away from achieving this vision.

However, for most natural resource systems, the level of benefits required to maintain human well-being have not been defined at the national or regional level, nor have the levels that might be desired in the future, which could act as focus against which decisions can be assessed.

Climate Change Act

The Climate Change Act (2008) sets out the requirement to reduce greenhouse gas emissions by 80%. This target has been set in response to the global environmental limit of no more than 2ºC of warming. The Climate Change Act used the best available scientific evidence to derive the target. The responsibility for deciding the most appropriate means of achieving that target, over what timescale and trajectory, was delegated to the independent Climate Change Committee. It is debatable whether the target chosen is the correct, given the scientific uncertainties involved, but research undertaken to establish whether the target will be achieved will inform whether the target is appropriate.

A key influence on how quickly the target will be achieved is the cost of carbon. It is known that the economic cost of damage from climate change is greater than zero. As scientific evidence of the impacts of climate change increases, this may lead to an increase in the cost of carbon, with a parallel rise in the value of natural capital assets that provide carbon sequestration services.

As with climate change, the other limits, such as the suggested global environmental limits associated with the planet's biophysical systems (Section 2.6), could be the basis of standards from which the gap in sustainability could be calculated. Targets to ensure that limits are not breached could be set in legislation, and mechanisms could then be put in place to promote the most appropriate means of achieving the targets over a given timescale.

As with carbon, the economic cost of the damage to natural systems will be greater than zero, so any cost attached to activities that impact on these boundaries is more correct than no cost. The revenues arising from charges for rising costs attached to polluting or damaging activities can be used to restore natural capital impacted by those activities, such as levels of biodiversity.

Early attempts to reflect this approach are seen in the UK's Aggregates Levy and Landfill Tax, both of which re-circulate gate fees, respectively for aggregate leaving a site and for landfill materials entering lanfill, for environmentally- and socially-beneficial purposes. However, a tighter focus on ecosystem services would need to see this kind of payment mechanism evolved into a true market that re-circulated the revenues from the costs of these activities into restorative or mitigating measures for ecosystems (Section 5.6).

4 The Ecosystem Approach and Environmental Limits

Overview

- Policy decisions at local and regional scales determine changes in benefit provision from natural resource systems and whether environmental limits are crossed which have consequences for future social wellbeing. Adoption of the ecosystem approach within existing decision making frameworks and methodologies would facilitate consideration of environmental limits.
- An evidence base is being developed to implement the ecosystem approach, a key part of which will be the ongoing UK National Ecosystem Assessment. This includes the impacts of consumption of ecosystem services arising from the transformation of natural resource systems both in the UK and in other countries, such as provisioning services for key commodities (for example timber, palm oil and soya).
- Ecosystem service valuation (ESV) offers the potential to place a value on the services forfeited by changes in use to balance against the economic benefits arising from any change. Prior to this, an ecosystem service valuation assessment (ESVA) is required to quantify the level of ecosystem services, including both capacity for ecosystem services to provide benefits and the degree to which these benefits are used.
- Valuation of changes can be incorporated into existing Cost Benefit Analysis (CBA) approaches used to assess the impact of policies. However, to ensure that the cumulative effect of many marginal losses of natural capital does not pose unacceptable risks, additional policy safeguards that incorporate environmental limits are required.

4.1 Ecosystem Management

Ecosystem management is the manipulation of the physical, chemical and biological processes which link organisms with their physical environment and the regulation of human actions to produce a desired ecosystem state.[114] The objective is to ensure the flow of multiple ecosystem services at levels acceptable for human wellbeing for both present and future generations by maintaining natural capital within environmental limits.

For example, the introduction of organisms to control invasive species, such as the non-native jumping plant lice (*Aphalara itadori*) being introduced in the UK to control Japanese knotweed, would constitute ecosystem management. In general, measures directly to manage ecosystem functioning can be carried out only on a limited scale, when understanding exists of how the processes, structures and functions in ecosystems will be affected and of the spatial and temporal extent of the likely effects (Section 3.2). In general, it is considered more practical to intervene to control human activities which impact ecosystem processes and structures, than to attempt to manage ecosystems directly.

Approaches that address human pressures on ecosystems are usually referred to as ecosystem based management. These attempt to manage pressures at the appropriate scale, such as a river catchment, taking account of the complex dynamics among organisms and their environment and how this developed and altered by human use. This does not require complete scientific understanding, but advice to policymakers on the basis of expert judgement, case histories and precautionary approaches to natural capital conservation.[114]

Applying Ecosystem Based Management

Ecosystem based management is applied through activities such as planning, regulation of damaging activities and economic or social incentives. However, the present UK and European legislative framework for nature conservation and land use planning was not designed to manage and maintain the flow of ecosystem services, and there has been little progress in incorporating relevant practices into the actual management programmes for natural resources to date. This failure stems not only from lack of market values and systems of economic analysis and accounting, but also from a limited understanding of ecosystems services.

On the principle that biodiversity should be protected for its intrinsic values to society, the majority of regulatory systems in place to protect biodiversity seek to maintain the ecological *status quo*, or if seeking to improve levels of biodiversity, to return an ecosystem to an historic benchmark. For example, the government set a target of returning designated SSSIs to “favourable condition” through appropriate measures. Given the extent of the current drivers of environmental change, it is unlikely that the ecological conditions to maintain current patterns of either biodiversity or ecosystem services can be maintained.

In addition, most SSSIs have been designated for their specific wildlife features, such as a particular species, and do not specifically protect ecosystem functions, structures and processes, such as hydrological processes, that support the features. The recent independent review, “Making Space for Nature” commissioned by Defra, concluded that “England’s collection of wildlife sites, diverse as it is, does not comprise a coherent and resilient ecological network even today, let alone one that is capable of coping with the challenge of climate change and other pressures".[57]

For example, the effects of climate change on biodiversity are already evident in the UK and, with any continued climate change, are expected to increase (POSTnotes 341). One key effect will be the changes in coastal areas due to rising sea levels (POSTnote 363). In the face of such changes, the possible future states of ecosystems and ecosystem services and the implications for social wellbeing need to be considered by policymakers, as well as how to identify alternative paths for reaching desirable states (Boxes 10 and 11).

Conservation policy has also previously focused on protection of areas of high species diversity, but it is not clear that these coincide with high levels of ecosystem services.[115] In line with the revised strategic plan of the CBD, the UK Biodiversity Partnership now puts greater emphasis on ecosystem services. This involves moving away from site and target based approaches to process based approaches to maintain the ecological integrity of natural resource systems. This requires a more complex evidence base as set out in previous chapters, but policy measures should reflect understanding of the systems involved and consider the utility value of biodiversity and ecosystem services.[116]

Trading Off Ecosystem Costs and Benefits in Decision-Making

To some extent, ecosystem management is a new focus to existing natural resource management activities, such as water management. It seeks to define ecosystem services for desired targets or to tackle specific problems, such as managing a river catchment to ensure that land uses do not impact on water quality and flood risk, while still delivering acceptable levels of other services such as food production.

Biodiversity conservation has been seen as a secondary objective to the delivery of societal and economic objectives such housing, transport, industrial production and agriculture. There are numerous different stable ecosystem states possible for a given location, each with different combinations of services and reflecting the different aspirations of those who could benefit or lose from changes in service delivery. Decision makers need to be aware of what aspects of well-being and which stakeholder groups are affected by changes to ecosystems (section 4.10).

Consumers of benefits are likely to vary geographically, socially and economically, with increased consumption of one service by one group having implications for the delivery of other services to different groups (Box 11). For example, the beneficiaries of ecosystem service provision, particularly provisioning services for commodities such as timber, palm oil and soya, are also different and distant from the places where ecosystem transformation occurs, leading to an institutional

Box 10 Regulatory Protection of Biodiversity and Coastal Realignment

The Essex Wildlife Trust created the largest coastal re-alignment in Europe in 2002 when over 200 acres of intertidal habitats were created by breaching 3.5 km of sea wall fronting Abbotts Hall Farm along the Blackwater estuary in Essex. Existing conservation designations for the site required the construction of a freshwater protection bund for an existing pond providing habitat for great crested newts (a protected species) and the construction of a new freshwater lake to compensate for the loss of ponds that were likely to become saline or brackish. This required the excavation of the lake site and the construction of a protective dam wall by raising the height of a farm track. The work necessary to obtain regulatory consent to create the saltmarsh cost more than the work undertaken to create the saltmarsh. The presence of protected species such as water voles, grass snakes and common lizards are likely in areas where managed realignment will be needed in future, requiring rescue programmes and habitat recreation, thereby significantly increasing costs.[117] However, the saltmarsh generated significantly more ecosystem service benefit provision than when the site was agricultural fields. Similarly, the Environment Agency maintains the drainage of large areas of arable land in Lincolnshire through intensive pumping infrastructure (diesel and electric based). This management approach does preserve some areas designated as SSSIs, justifying the drainage, but allowing water levels to rise and flooding to occur would result in the restoration of the once extensive areas of traditionally-managed grazing marshes and dykes (Lincolnshire Coastal Grazing Marshes Project).

failure to address the drivers of natural capital consumption.[118] In addition, current changes in some benefits, such as biodiversity, have implications for future consumers of ecosystem service benefits (section 4.9).[116]

4.2 The Ecosystem Approach

The ecosystem approach makes explicit the link between the status of natural resource systems and human well-being. It is a form of ecosystem based management that seeks to maintain the integrity of ecosystem functioning and to avoid rapid undesirable ecological change. As ecosystems are a conceptual unity (Box 1), management measures seek to influence chemical, hydrological and biological factors to maintain environmental conditions, rather than managing ecosystems as a separate entities.

The objective of the ecosystem approach is to ensure that governance mechanisms balance levels of use of natural capital with its conservation. Policies should also reflect that the impacts of human activities are now an integral part of ecosystem interactions, just as ecosystems are to human activities. The approach also requires that policies are informed by an understanding that the affected processes underlying ecosystem services are inherently complex and dynamic at many temporal and spatial scales. For example, some types of fishing activities can alter marine ecosystem processes on a large scale, such as nutrient cycling, and equally the collapse of fish stocks can affect social and economic systems.

Agenda 21, developed at the Earth Summit in Rio de Janeiro in 1994, stated that integrated management of natural resources is the key to maintaining ecosystems and the essential services that they provide. The Sibthorp Seminar held in the UK in 1996 defined ten principles of ecosystem management, five guiding principles:[119]

- management objectives are a matter of social choice;
- ecosystems must be managed in a human context;
- ecosystems must be managed within natural limits;
- management must recognise that change is inevitable;
- ecosystem management must be undertaken at the appropriate scale and conservation must use the full range of protected areas;

and five operational principles:

- ecosystem management needs to think globally but act locally;
- ecosystem management must seek to maintain or enhance ecosystem structure and functioning;
- decisions makers should use appropriate tools derived from science;
- managers must act with caution; and,
- a multidisciplinary approach is needed.

These principles were elaborated to form the basis of the Ecosystem Approach adopted by Convention on Biological Diversity (CBD) in 2000 as 12 complementary and interlinked principles (Set out in Annex C). The CBD describes the ecosystem approach as "a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way". The broad scope of the principles goes beyond ecosystems themselves, and encompasses social, cultural and economic factors that are fully interdependent with biodiversity and ecosystem services.

The Ecosystem Approach and Environmental Limits

Implementation of the ecosystem approach for management of activities affecting ecosystems could be the basis of a systematic definition of environmental limits, defined at the national, regional and local level, and ensuring that trade-offs between natural and other capital do not lead to them being exceeded. The ecosystem approach is widely recognised as an appropriate conceptual framework within which environmental limits to natural resource use can be identified and used in decision-making.[120]

Principle 6 of the ecosystem approach explicitly states that management of ecosystems should take a precautionary approach "to the environmental conditions that limit natural productivity, ecosystem structure, functioning and diversity". However, as stated in previous chapters, there are difficulties in defining when ecosystems are near the limits of the environmental conditions that maintain their character, as well as the environmental tolerances and requirements of organisms associated with these ecosystems which are critical to their functioning.

Implementing the Ecosystem Approach in England

The development of the evidence base to implement the ecosystem approach in England is ongoing. Policies with environmental impacts on natural resource systems are implemented by a range of organisations with differing priorities, including government departments, creating a significant constraint in delivering the multiple objectives of the ecosystem approach.

In December 2007, Defra published "Securing a Healthy Natural Environment: an action plan for embedding an ecosystems approach". This was updated in the 2010 "Delivering a Healthy Natural Environment". The guidelines are intended to advance how environmental values and ecosystem services are taken into account in policy decisions. Supplementary guidance to the Treasury's Green Book on accounting for environmental impacts in policy and project appraisal will also be published.

There is considerable complexity in understanding and assessing the causal links between a policy, its effects on natural resource systems and related ecosystem services and then valuing the effects in economic terms. In particular, land use systems are complex. This includes the characteristics arising from the natural interactions that give rise to soils, topography, hydrology and biodiversity in a landscape, the human interactions that use the benefits from the natural resources and the governance institutions that regulate these interactions.

The recent Foresight report on land use futures stated that new frameworks are required to shift the current policy focus of immediate environmental damage as a cost incurred to processes that systematically capture the full range of wider impacts on ecosystems and human wellbeing now and in the future (Figure 8).[121] Defra's emphasis on "an ecosystems approach" rather than "the ecosystem approach" is designed to avoid rigid interpretation of the CBD principles (Annex C) that may not be relevant in all circumstances to all stakeholders. Defra is seeking to apply a generic set of principles to policy making and delivery, including:

- Taking a more holistic approach to policy-making and delivery, with the focus on maintaining healthy ecosystems and ecosystem services.
- Ensuring that the value of ecosystem services is fully reflected in decision-making.
- Ensuring environmental limits are respected in the context of sustainable development, taking into account ecosystem functioning.
- Taking decisions at the appropriate spatial scale while recognising the cumulative impacts of decisions.
- Applying adaptive management of the natural environment to respond to changing pressures, including climate change.

The onus of the Defra approach is to maintain ecosystem service benefits at levels that maintain human well-being, in line with the 'quality of life' requirement of the current Defra Business Plan (Structural Reform Priority 2). This is subtly different from the CBD ecosystem approach which sees humans as an intrinsic part of ecosystems and has a greater focus on bottom up approaches that enhance the immediate wellbeing of local communities and ecosystem service benefit users as these stakeholders perceive it.

Winners and Losers

Management of ecosystems to enhance benefits for a wider regional set of stakeholders may be at the expense of local stakeholders. For example, coastal realignment in estuaries to protect key urban areas are implemented in rural areas, where residents may perceive unfavourable effects of such changes on ecosystem services benefited from locally (Box 11). Similarly, decisions favouring local stakeholders can have cumulative impacts at national and regional levels, such as land management decisions in upstream areas which can affect flood risk and water quality downstream (Box 14).

The Defra approach could be implemented as a top down 'black box' approach, which mandates how ecosystem service benefits should be assessed and valued. However, given the inherent complexity and interconnectedness of ecosystems, the ecosystem services most enhanced may be those that are most easily monitored and that lend themselves to target setting and national accounting systems, given the inherent complexity and interconnectedness of ecosystems. This could lead to perverse decisions, both in terms of subsidies or land use change, or the perception that ecosystem service considerations are an additional bureaucratic burden at the local level, with benefits flowing elsewhere. To address this issue, guidelines for stakeholder participation in decision making processes have been introduced (Section 4.12).

Environmental limits can be used to focus decision making on flows of benefits that are particularly critical to human well-being and where economic valuation (Section 4.10) is likely to be insufficient to deal with the level of risk incurred. Although a strategic top-down approach may be the most effective means of implementing precautionary environmental limits, there is the potential for unintended consequences. There are inherent tensions between national and regional decisions to avoid the breaching of environmental limits and the subsequent constraint on decisions at the local scale, raising questions about the level at which environmental limits should be agreed and implemented.

4.3 UK National Ecosystem Assessment

In a recent report, the House of Commons Environmental Audit Committee called for an over-arching ecosystem assessment of the UK at the national level.[122]

Box 11 The Alkborough Flats Managed Realignment Scheme

Coastal change occurs through the action of wave, wind and tide processes. In areas where these are highly dynamic and 'energetic', only the most resilient geological materials will remain. Soft and low lying coastlines and estuaries are continually shaped by these processes, which cause land to be removed and the material to be deposited out to sea or in coastal areas where the processes are less 'energetic'. For over 300 years, UK management of coastal lowlands has been dominated by land reclamation and flood protection, principally through the construction of sea walls and drainage of wetlands. However, habitats found in soft and low-lying coastlines and estuaries, such as mudflat, sandflat and saltmarsh are likely to decline due to coastal squeeze between rising sea levels and sea defences or roads and because of unregulated land-use change.[123] On the Essex coastline, medieval to 19th century embankments have caused the historical loss of 40,000ha of saltmarsh.[124] Loss of intertidal habitat may increase the wave energy reaching engineered coastal defences and cause maintenance costs to escalate (POSTnote 342).

The need to create accommodation space for sea level rise (POSTnote 363) has led to managed coastal realignment becoming the preferred option for managing flood risk. This involves breaching sea walls to allow the sea to cover uninhabited land as far inland as the nearest high ground or new sea walls. The land affected would usually have been saltmarsh before the original defences were created to provide agricultural land.[117] The Humber is a major estuary draining one fifth of the land area of England. Alkborough Flats is located on the south bank of the inner Humber estuary at the confluence of the Rivers Ouse and Trent. On the landward side of the flats is a natural escarpment, making them a good location for coastal realignment. This was designed to deliver flood risk management and biodiversity benefits as well as social and economic benefits to the local community. It constitutes part of a wider Humber Flood Risk Management Strategy that aims to protect the homes and businesses of over 40,000 people. By allowing the 440ha Alkborough Flats to flood, high water levels are reduced over a large part of the upper Humber estuary by 150mm. At a projected annual sea level rise of 4mm per year until 2025, and then 8.5mm per year until 2055, this should continue to reduce high water levels for another 25 years and make it possible to defer the building of flood defences upstream.[125]

170ha of the site are exposed to flooding, reverting to mudflat, saltmarsh and in parts reedbeds, with the remaining 230ha acting as flood storage during extreme surge events and primarily used for grazing. The section permanently exposed to flooding now supports a wide range of wildlife including waders and other birds. Extensive tidal mudflats make the Humber Estuary important for wildlife, including over 160,000 waterfowl annually. Over 3,000ha of intertidal habitat has been lost in the mid-outer estuary since 1850.[126] Other estuary habitats of importance for biodiversity include sand bars, shingle banks, saltmarsh, saline lagoons, reedbeds and freshwater marshes. The total Humber estuary includes seven Sites of Special Scientific Interest (SSSIs), is a designated Special Protection Area under the EC Birds Directive, and contains a range of Special Areas for Conservation under the EU Habitats Directive (SACs). Much of this habitat may be lost as a result of sea level rise in the next 100 years, and longer-term maintenance of biodiversity will depend on allowing the estuary to change in response to sea level rise.[125]

The ecosystem service benefits arising from the realignment were assessed and economic values for non-marketed benefits were derived where the benefit could be quantified. The cost-benefits appraisal should include the range of ecosystem service benefits in addition to the flood defence costs.[127] The baseline used for this was the former intensively arable farming across the site. It had initially been assumed that benefits from provisioning services (e.g. food production) would be reduced in favour of regulating services (e.g. flood regulation), supporting services (e.g. biodiversity) and cultural services (e.g. recreation). However, despite a number of uncertainties, the assessment estimated that the change in land use was neutral or slightly positive for provisioning services, with the value from wool and meat from rare breeds grazing of sheep and cattle offsetting loss of arable production of food and fibre. It is likely that commercially-exploited fish species using the saltmarsh as a nursery area add substantial extra value, but methodological shortfalls prevented their valuation. Overall, the assessment found a significant improvement in ecosystem service benefits arising from improved ecosystem functioning.[125]

The development of adaptable and resilient coastal zones, which include both resilient ecological and social systems (Section 5.2), remains a major policy challenge in the UK (POSTnote 342). Any proposal to change existing coastal defences is likely to cause concern among coastal communities, with many stakeholders regarding realignment as giving up land to the sea. Communities in areas where benefits might be gained from realignment schemes hold strong views about how their surroundings should be managed and can object to adapting areas to environmental change. There are substantial difficulties in conveying ecosystem service benefits to stakeholders when there is a general perception that changes in current land use are a loss and where the primary beneficiaries of these schemes are geographically remote. Although land is purchased prior to flooding, the loss of high grade agricultural land is a major local issue. The flood risk maybe reduced by the realignment, but the reduction in distance between settlements and the river is perceived as increasing vulnerability. Concerns arise from disparate interests, from shellfisheries, navigation, and public access and to protected biodiversity. Some also fear that that the creation of wetlands will increase the prevalence of disease vectors such as the mosquitoes that are vectors for malaria or the midge flies that transmit bluetongue, although saltmarsh would not provide suitable habitat for these species. A planning application for a similar realignment scheme under the Humber Flood Risk Strategy at Donna Nook on the North Lincolnshire Coastline has attracted significant opposition, including from local MPs and MEPs.[125]

Coastal community groups have objected to the economic valuation of environmental and cultural assets to determine the costs and benefits of their loss (Section 4.8), as they believe that the loss of these assets is fundamentally unacceptable and that the valuation system is inherently biased against rural communities (POSTnote 342). With realignment in the Humber estuary, the main economic value of flood risk reduction benefits accrues mainly to the more densely populated urban areas, whereas enhanced hard defences of agriculture land would be a cost to the wider public with a reduction in ecosystem service benefits to that public. If coastal realignment is more widely applied as part of climate change adaptation strategies, large tracts of high quality agriculture land and some housing are likely to be 'sacrificed' and friction over the subsequent social justice and compensation issues will increase. The ecosystem approach offers one means by which decision makers could transparently trade-off loss of local benefits against environmental limits for regional coastal flooding.

To this end, Defra, the devolved administrations, Natural Environment Research Council (NERC) and the Economic and Social Research Council (ESRC) have undertaken an ecosystem assessment for the UK based on the principles developed in the MA and articulated in the World Conservation Monitoring Centre (WCMC) 2009 "Manual on Ecosystem Assessment."

Part of the Living With Environmental Change (LWEC) initiative, the assessment began in mid-2009 and will be reporting its findings in March 2011 (Box 12). The National Ecosystem Assessment (NEA) has collated and synthesised existing evidence on the UK natural environment to give a comprehensive picture of its current state and the provision of ecosystem services, and has explored scenarios for future change, from the present day up to 2060.[128] Previous global scenarios of changes in biodiversity for the year 2100 identified that, for terrestrial ecosystems, there will be continued degradation, mainly caused by land use changes, with climate change, nitrogen deposition and invasive species having substantial impacts.[129]

The NEA is intended to act as a natural capital asset check to determine whether current UK policies and projects have a positive or negative impact on natural capital critical to maintaining human wellbeing (Box 12).[130] An inventory of ecosystem services appropriate for UK country and regional scales context is a prerequisite for developing indicators to reflect possible alternative ecosystem states resulting from decisions (Annex B).

The NEA assessment is intended to create a compelling and easily-understood explanation of the state and value of the UK's natural environment and ecosystem services. Rather than seeing environmental protection of the UK environment as a cost, it communicates how UK ecosystems and ecosystem services are an "integral part of society". An assessment of the pressures acting on ecosystems will give the context for changes in the states of ecosystems and the delivery of ecosystem services that stem from those ecosystem states.

The evidence base to determine a baseline for ecosystem service delivery in the UK has, to date, been sparse at best. The NEA will help to inform the qualitative assessment of the future potential impacts of policy options on the provision of ecosystem service benefits (Annex D). Once the initial stages of taking stock and organising knowledge about ecosystem service provision has been achieved, this information can be used to:

- facilitate communication and discussion of management scenarios with stakeholders in ecosystem service benefit provision;
- investigate different weights and values for different outcomes;
- help users of the information to make links across the diverse topics involved in reaching the decision; and,
- inform the design of policies, such as payments for ecosystem services or agri-environment schemes (5.8).

Preliminary findings of the NEA have suggested that:[131]

- Significant changes have occurred in broad habitat types (Box 12) throughout the UK since the Second World War, driven by economic growth, demographic changes, advances in science and technology, government policies and individual behaviours.
- Significant biodiversity loss has been documented in the UK in the last 50 years, but data for many taxonomic groups, including many insects and microbial communities are poor, creating difficulties in comprehensively assessing ecosystem status and trends.
- There have been significant increases in levels of provisioning services (such as agriculture and forestry), but this has been at the expense of other types of ecosystem service benefits such as some regulating, supporting and cultural ecosystem services (Figure 9).

4.4 Evaluating the Impact of Policy Changes on Ecosystems

The value of the natural resource systems is the value of the flow of benefits less the cost incurred to produce those benefits. Some of these have direct market values, such as crops grown, whereas others such as the regulation of water flows by wetlands do not. Although the appraisal of new policies already requires environmental impacts to be taken into account,[132] the methodologies used do not recognise the costs of ecosystem degradation and the benefits from better management of ecosystems.

Several recent frameworks, such as the Defra (2007) introductory guide to valuing ecosystem services, have suggested the required approach for evaluating the cost of impacts includes:

- Establishing the environmental baseline (describing the habitats present and the ecological processes they support).
- Identifying and providing qualitative assessment of the potential impacts of policy options on ecosystem services, through an

ecosystem service valuation assessment (ESVA);
- Quantifying the impacts of policy options on specific ecosystem services and assessing the effects on human welfare.
- Valuing the changes in ecosystem services, through an ecosystem service valuation (ESV).

These steps are intended to provide a framework for a systematic approach to accounting for impacts on ecosystems. Similar frameworks have been proposed by TEEB and the US Environmental Protection Agency Science Advisory Board. There is general consensus that there should be an initial assessment phase, followed by more detailed biophysical, social and valuation assessments.[133] However, even an initial assessment of ecosystem services affected by a policy choice can indicate how potentially significant impacts could be and where uncertainties and evidence gaps lie.

The first step in establishing an environmental baseline to identify and categorise ecosystems and their services impacted by policy options. This will characterise the nature and extent of an ecosystem, such as intertidal saltmarsh (Box 11), its interdependence with other ecosystems, such as marine ecosystems, and services provided, such as coastal protection, carbon storage, recreational uses, maintaining water quality and acting as a nursery area for commercially exploited marine species.

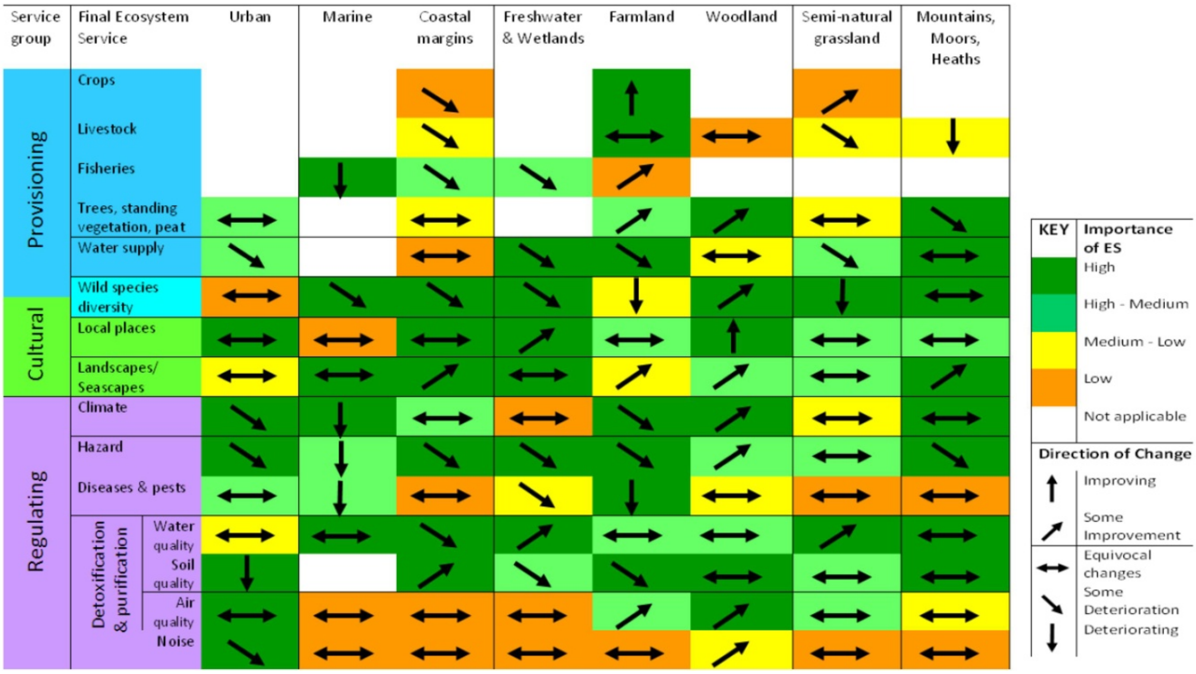
Box 12 The UK National Ecosystem Assessment (NEA)

The UNEP-World Conservation Monitoring Centre (UNEP- WCMC) based in Cambridge was appointed to co-ordinate and provide a secretariat for the NEA in February 2009. The NEA entailed a first phase to join up the evidence base and provide a high level assessment of the status and trends of ecosystems and ecosystem services, drivers of change for ecosystems and ecosystem services, valuation of services and links to human well being, which was published as an interim report. The UK NEA grouped BAP Broad Habitat types into eight more general units covering terrestrial, freshwater and marine ecosystems:
- mountain, moors and heaths
- semi-natural grassland
- enclosed farmland (consisting of arable, horticultural land and improved grassland)
- woodland (covering broadleaved, coniferous and mixed woodland)
- freshwater, wetlands and floodplains
- urban
- marine
- coastal margins.

While the habitat approach to service assessment, as opposed to an approach based on Service Providing Units (SPUs) or Ecosystem Service Providers (ESPs) (Section 2.4), has a number of advantages, it might not be able to provide a picture of what is happening to individual services. For example, the contributions individual habitats make to aggregated assessment of ecosystem service output is often unclear and weighting habitats by their area may not reflect these contributions. However, the broad habitat types are similar to those used for other habitat classification systems, such as the Countryside Survey.[134] The NEA did not undertake new analyses but drew on available published results . The outputs from survey and monitoring schemes, state of environment reports, progress reports and various research projects were incorporated, including:
- the Countryside Survey and its various component projects such as the integrated assessment of ecosystem services [135]
- the BICCONET project that is collating evidence of climate change impacts on biodiversity (POSTnote 341)
- the environmental monitoring work contributing to the UK Environmental Research Funders Forum (ERFF)
- the Environmental Observation Framework (EOF)
- work on valuation of ecosystem services
- various initiatives to promote application of the ecosystem approach.

The second phase, from February 2010 to February 2011, looked at the future, and how the ecosystems and services are likely to change under a number of scenarios, an analysis of UK's dependence on non-UK derived ecosystem services, and possible options to deal with uncertainties and to safeguard ecosystem service benefits. Six scenarios were used to explore how ecosystems and their services in the UK might change in the future (up to 2050), and to identify what the possible effects might be in terms of human well-being and the parties likely to be affected.[136] As well as providing an overview for the whole of the UK, the assessment, where possible, was broken down to country level and regional levels to determine:
- the status and trends of the UK's ecosystems and the services they provide to society
- the factors causing ecosystem changes in the UK
- how ecosystem changes have affected human well-being in the UK
- the beneficiaries of ecosystem services in the UK, and their location. considering trade-offs between types of services
- how the location of beneficiaries of ecosystem services affects how the ecosystem services are valued and managed
- possible negative effects of changes in ecosystems on society and who might be most affected
- how ecosystems and their services in the UK might change in the future under plausible scenarios
- high-level policy responses that may be appropriate to secure continued delivery of UK ecosystem services under plausible future scenarios
- the key ecosystem services upon which the UK depends that are not provided by UK ecosystems
- the policy implications of UK-dependence on non-UK ecosystems
- the uncertainties, and knowledge and data gaps for understanding and managing the supply of ecosystem services in the UK, including risks of sudden change, trade-offs between beneficiaries, and impact of policy responses.

Figure 9 Preliminary NEA Analysis



Preliminary NEA analysis of the relative importance of different habitats (coloured cells), and change over the last 20 years within them (directional arrows), in delivering final ecosystem services across the UK. It will differ at different national, regional and local scales and is likely to be subject to revision before completion of the NEA

The establishment of environmental limits is complementary to other aspects of implementing the ecosystem approach as well as integral to it. This includes provision of adequate baseline information on natural resource systems and related ecosystem services: the environmental state of the system; the pressures acting on the system, the benefits provided by the system; the relationship between pressures, states and levels of benefit provision; and, how the system behaves (Box 13). However, given the uncertainties in this information for most ecosystem services, the degree to which they can be set out quantitatively in regulation is questionable and environmental limits may need to be considered qualitatively within ecosystem management frameworks.

4.5 Scale-specific Decision Making

The ecosystem approach requires open and explicit choices to be made between alternative stable ecosystem states and the levels of benefits they deliver, but the current decision making framework does not yet do this at relevant local scales. Any land parcel area can have multiple uses and values given to it by different actors, requiring trade-offs amongst different actors.

Making such choices would require decision makers, landowners and other stakeholders to work together to achieve commonly agreed ecosystem goals in a given locality. Changes in the way ecosystems are managed are always likely to be contentious because they alter the incidence of both beneficial and adverse consequences, whether these are financial or in kind. This not only leads to conflict over private gains versus wider public benefits but also about who gains benefits and who bears costs. Not all of these issues will be resolved through approaches such as public participation (Section 4.9).

Box 13 Incorporating Environmental Limits

The general process for establishing environmental limits can be summarised as[18]:

- Establish the range of benefits provided by different natural resource systems and the potential for the provision of multiple benefits and trade-offs between those benefits as a result of different policy options (Sections 4.5).
- Establish an evidence base for suite of relevant indicators to monitor the delivery of benefits from a natural resource system against environmental limits set, including evidence of biophysical thresholds (Section 2.7).
- Set environmental limits on the basis of both scientific evidence and societal considerations as to the level at which benefit provision becomes unacceptable. Where there is scientific uncertainty as to where biophysical thresholds lie, the level of risk that communities are willing to incur becomes more critical.
- Implement management measures to ensure that the level of benefit provision is maintained above environmental limits. The extent of management measures implemented should reflect either the economic value gained from maintaining benefit provision above environmental limits (Section 4.6) or the resilience of the natural resource system in question (Chapter 5).

The relevant scale for mapping ecosystem services remains a matter of debate as ecosystem management decisions can be taken at the national, regional or local level. Local scale areas are based on landscape boundaries, such as an area of similar geology like chalk downland or a river catchment. The size of the area should not only be ecologically relevant, but also socially, economically and culturally appropriate, such as National Park Authority areas. However, ecological scales do not usually match decision making scales, creating difficulties for assessments, valuation and limits.[137]

The spatial layout of ecosystems and the natural capital stock within them is important for the interactions that give rise to beneficial processes and ecosystem services. For example, ground, surface and precipitated water are all linked across entire catchment areas and impacts on any one aspect will affect hydrological processes within the catchment and the ecosystem services arising from these, such as drinking water quality. Equally, the social value of ecosystem services relates spatially to where they are consumed. Most natural capital stocks, such as a wetland, cannot be transported to another location, meaning that some ecosystem services are location specific, particularly regulatory, supporting and cultural ecosystem services (Figure 3).

For example, a Marine Conservation Zone (MCZ) Project (POSTnote 310) has been established by Defra, Natural England and the Joint Nature Conservation Committee (JNCC), to identify and recommend MCZs to Government. The project is being delivered through four regional initiatives covering the south-west, Irish Sea, North Sea and the south-east. The south-west MCZ project (called the Finding Sanctuary Project) not only includes biological and physical information about the sea areas, but also the intensity and patterns of use of sea areas by different stakeholder groups to construct spatially and temporally explicit mapping of the supply and demand for ecosystem services.[138]

A full spatial classification of ecosystem services and their quantification and mapping for each location would also take into account local, regional and global consumers of ecosystem service benefits. This allows ecosystem service flows between different regions to be determined, and the identification of areas or stakeholders obtaining the benefits provided by ecosystem services, as well as areas or stakeholders incurring a loss of service provision. However, the extent to which this is necessary is dependent on the policy or management change being considered and the scale of the likely impacts. The development of spatially explicit ecosystem service indicators at appropriate scales is critical to assessing impacts (Annex B).

Quantitative Assessments of Ecosystem Service Provision

The performance and availability of ecosystem services have to be quantified by some direct measure, including capacity to provide benefits and the degree to which these benefits are used. Changes in the delivery of benefits to humans under different possible scenarios can then be estimated and physically mapped to inform policy makers about the impacts of any future management choices. This requires:

- a sufficient understanding of the factors that are key to the production of benefits;
- how they are likely to be affected by any possible changes;
- spatially explicit mapping to understand where benefits arise and are used, as well as the variability of costs of conserving benefits in different locations; and,
- spatially explicit information on benefits that can be used to identify environmental limits at different spatial scales, including the national, regional or local.

Most published assessments attempt to map services, their flows and the external pressures on them,[139] and the development of methods is the focus of ongoing research efforts, such as 'The Natural Capital Project' in the USA.[140] A systematic characterisation of ecosystem services such as provisioning services in biological and physical terms means, for example, quantifying the flows of goods harvested in an ecosystem in physical units. Physical mapping could use techniques such as Geographic Information System (GIS) tools – computer based systems designed to store, retrieve, analyse, model and map large volumes of spatial data. Layers of biodiversity and ecosystem data could be overlain on maps to reveal key areas for habitat protection, management and restoration in relation to ecosystem services. Relevant datasets need to be drawn from a wide range of providers, including information such as species records, habitat boundaries and geographic boundaries.

This information can be overlaid onto other social and economic spatial data sets to characterise the likely impacts of developments for residential, commercial or transport purposes. Reviews have concluded that mapping approaches to ecosystem services provide a good basis for developing a systematic planning framework that offers the scope for informing trade-offs, as well as synergies between different objectives. However, mapping studies have been criticised for being overly concerned with natural capital, such as habitats and rarely considering markets or beneficiaries.[85]

In most cases, there will be sufficient evidence to determine if decisions will positively or negatively change ecosystem service levels, such as the reduction of peak flows downstream through land cover changes (Box 14). An important component of such approaches is the development of dynamic models of landscape change to allow the spatially explicit depiction of the impacts of alternative land use and management options, such as using farmland for flood storage (Box 15 and Figure 10).

For example, the INnVEST suite of models developed for the US Natural Capital Project can be used to explore the consequences of different options and choices with stakeholders, providing outputs in the form of maps, trade-off curves and balance sheets. [141] Such scenario modelling provides information on potential ecosystem services for each location and identifies possible synergies and likely trade-offs in ecosystem service benefit provision. This has been done at the national level for some ecosystem services for the NEA Scenarios. However, cultural services such as sites valued for religious reasons, cannot be easily expressed in quantitative units as they are dependent on human interpretation.

Once a spatial assessment of the condition or state of an ecosystem has provided an environmental baseline, and there is an understanding of how that system might change under different sets of policy or management interventions, the impact of other drivers of environmental change, such as climate change should be considered. Such approaches could target where beneficial land uses should be incentivised at the landscape level to ensure ecosystem services are sustained within environmental limits, such as habitat restoration to reduce habitat fragmentation, and identify where specific types of land use, such as drainage of wetlands, would be detrimental and so should be discouraged.

4.6 Ecosystem Valuation

There is a growing body of academic literature on valuation exercises to frame and quantify the economic costs and benefits of changes in biodiversity and ecosystem services (Section 3.4). Ecosystem services are paid for when any benefit from provisioning services is traded (food, timber etc.), although these are classed as ecosystem goods by economists. An ecosystem good is tangible material product resulting from ecosystem processes, whereas ecosystem services are an improvement in the condition or location of things of value (such as ecosystem goods).[142]

For example, if a tree sapling is sold, it could be grown to harvest the timber, grown to offset the carbon emissions of the purchaser or grown for the appreciation of its aesthetic beauty in a garden. For the first of these options, the benefit is realised as an ecosystem good; for the second, the benefit is derived via regulating ecosystem services; for the third, the primary benefit is in terms of cultural ecosystem services (Box 16). Some benefits arising such as recreational opportunities from improving the garden are less straightforward to categorise.

Apart from provisioning services, there are no direct markets for services with the exception of pollination services (Box 2) through the purchase of hives by growers, but there are indirect markets such as the carbon market and tourism. For the most part, decision making processes take account only of the market price of land or the value of crops it will produce, but ignore the value of the majority of ecosystem services that will be altered by land use change. The valuation of ecosystem service benefits offers the potential to place a value on the services forfeited by changes in use, to balance against the economic benefits arising from any change.

The value placed on different benefits from ecosystem services can alter the way ecosystems are managed. The economic valuation of ecosystem services can also serve multiple purposes beyond informing decision makers, including:

- increasing awareness and understanding of the actual and potential service benefits for human wellbeing;
- facilitating communication regarding these benefits with different stakeholders and the general public;
- collating and processing information about the impacts of management options, to inform priorities;
- expression of impacts in monetary units, commensurable with other economic effects;
- identifying which impacts are included and which are not in the estimates, and avoiding double-counting; and,
- informing policy instruments that directly reflect environmental impacts, such as the Landfill Tax and Aggregates Levy.

Robust Valuations

Although processes for determining robust values for natural resources and ecosystem services are ongoing (Box 16), reports, such as the TEEB report for policymakers, contend that there is already sufficient information for values to be incorporated into decision making processes. As part of the UK National Ecosystem Assessment (NEA), derived values for quantified benefits from ecosystem service benefits provision will be physically mapped.

Economic valuation frameworks usually place values on the outcomes which directly support human wellbeing (Section 3.2), as is the case with UK NEA framework (Figure 11B). However, valuation frameworks also need to define the processes that directly give rise to a benefit, to understand its spatial distribution and where the benefits of these processes are realised.

Box 14 Land Use, Flood Risk and Water Quality.

Approximately 1500km^2 of blanket peat in upland Britain is drained by grips, open steep sided drainage ditches, to lower water tables to improve vegetation for grazing and grouse. Hydrological modelling studies indicate that this increased rate of drainage, in comparison with intact blanket bogs, increases flood risk downstream as well as impacting water quality by increasing rates of peat breakdown or erosion. Studies from lowland areas indicate that high livestocking densities also cause soil compaction, reducing infiltration and enhancing surface run off increasing flood risk, although studies in upland areas are limited.[143] UK peatlands include globally rare habitats and constitute the largest terrestrial store of carbon in the UK and are also important for good water quality and flood control. As the UK climate changes in coming decades, models suggest that more than 50% of UK peat will be vulnerable to change by 2050. Climate change will exacerbate current pressures, such as drainage, wildfires and grazing, increasing erosion rates with catchment scale impacts on flood regulation and water quality.[144] However, at present there is very little evidence that enhanced surface runoff resulting from land management practices reduce flooding at the catchment scale, although this lack of evidence does not imply this has not taken place.

United Utilities, under their Sustainable Catchment Management Plan (UU SCaMP), are implementing restoration within the upper catchment of the River Hodder in the Forest of Bowland. This is targeted primarily towards improving water quality and the status of the ecological quality of upland mire habitats, which include 13,500ha designated as Sites of Special Scientific Interest (SSSI) and part of the 16,000hae Bowland Fells Special Protection Area (SPA). A key water quality issue is minimising processes that generate colour in the waters, principally through breakdown and erosion of peat which releases organic acids. These acids can react with chlorine during water treatment to form trihalomethanes, which may have adverse health effects and are regulated (along with colour) in drinking water. The main land management measures instituted are:

- blocking gullies or grips to increase water levels and reduce the loss of peat to downstream reservoirs in around 5,500ha of blanket peat bogs;
- reduction or relocation of sheep grazing, to restore vegetation and reduce erosion;
- planting of deciduous woodland trees, to provide bird habitat and to improve slope stability in river corridors, comprising about 300,000 trees in 450ha;
- controls on heather burning;
- bracken control to restore mire plant communities; and
- pond (scrape) creation for wading bird habitat.

Around 96% of the SSSIs in the SCaMP area are now in favourable or recovering condition 294ha of woodland is already planted, 33 km of moorland grip blocked and 60ha of exposed peat restoration underway.[145] The implementation of land management changes across a catchment provides an opportunity to analyse its effects on downstream flooding. From a flood generation perspective, the upland management measures that are most likely to affect flood generation directly are grip blocking, changes in stocking density, woodland planting and scrapes. An Environment Agency funded study, 'Multiscale Experimentation, Monitoring and Analysis of Long Term Land Use Change and Flood Risk', is attempting to address this lack of data through a study of the SCaMP area. The study is intended to ascertain the impact of management measures through an intensive water flow monitoring programme at a range of different scales as management measures are implemented across the catchment.

Despite the multiple ecosystem service benefits of naturally functioning flood plains and wetlands, many have been converted to support uses that provide other specific benefits (Box 15 and Figure 10).[146] A €3.8 million (about £3.4 million) project, to improve the quality of river water flowing into the English Channel, is being led by the Westcountry Rivers Trust funded under the sustainable management of natural resource strand of EU Interreg IVA, France (Channel) England programme. Partners include South West Water, the Environment Agency, the Association of Rivers Trusts and six French environmental, commercial and local government organisations. The WATER project (Wetted Land: the Assessment, Techniques and Economics of Restoration) aims to create and permanently protect wetland within a river catchment. Farmers will be funded to take small areas of riverbank and wetland, which have limited food production use, out of cultivation. This is intended to reduce the risk of downstream flooding, boost fish numbers, improve wildlife habitat and improve water quality. It also links to existing initiatives including the Exmoor Mires Restoration Project, which uses the natural filtering effect of peat to deliver cleaner raw water to water treatment works and reservoirs, reducing the need for future expensive additional storage capacity or energy-intensive treatment plans. The Environment Agency and OFWAT are having ongoing discussions about the costs of the impacts of agriculture on water quality as the basis for the expansion of such water company funded schemes.

South West Water are investing almost £9m in their "Upstream Thinking" initiative, which aims to provide clean water through helping landowners to choose farming methods that enhance water quality while also protecting natural resources and improving the quality of wildlife habitats. The investment made should reduce requirements for energy intensive water treatments and it has been calculated by South West Water that the benefit-to-cost ratio is over 65:1. The scheme will include investing in the Working Wetlands project, a 7 year Devon Wildlife Trust initiative focussed on 65,000ha of land which contains the headwaters and main tributaries of four major rivers (Taw, Torridge, Exe and Tamar). This aims to help farmers to maintain, restore or recreate wet grazing pasture (called culm grassland). Since 2008, when the project started, over 700ha of this has been brought into recovering or favourable condition, with its restoration on over 73ha of former Sitka spruce plantation and 50ha of improved grassland, as well as 45km of riverside land management changes.[57]

Box 15 Flood Storage

The Government's strategy "Making Space for Water" includes reconnecting rivers with floodplains to provide natural flood storage. In England, over one million hectares of agricultural land lie within the indicative floodplain, that is, has an annual risk of flooding from rivers of 1% or greater or from coastal flooding of 0.5% or greater. Although this accounts for only 9% of the total agricultural area, it includes some of the most fertile and productive areas that have been 'reclaimed' and 'improved' for agricultural purposes over many years. The agricultural productivity of this land is maintained by the management of hydrological regimes in the form of flood alleviation and land drainage. [147]

Managing this area for flood storage would be beneficial for downstream, often urban, areas but involves costs for rural areas. A survey of farms affected by the severe summer 2007 floods showed an average of almost £1,200 per hectare flooded and a total cost of about £50 million on 42,000ha of agricultural land flooded. [148] There is also a possible conflict between biodiversity objectives and maximising flood storage, as the creation and maintenance of some types of wetlands require that the water table in an area is continually high, reducing available storage capacity. Managing these areas to deliver an array of ecosystem services is possible if trade-offs are made between different benefits (Figure 10). For example, locally relevant and targeted agri-environment options (Section 5.8) can help to balance production and environmental protection, and may be able to offer the greatest combined output of ecosystem goods and services. However, as the Alkborough Flats case study (Box 11) showed, innovative land uses may avert 'trade-offs' and yield benefits across ecosystem service categories. Given that different benefits will be important to different stakeholders, an ecosystem assessment would need to:[147]

- identify and quantify the range of services provided by floodplains under different management options;
- determine how benefits and costs are distributed among different stakeholder groups, facilitating dialogue among them and showing what can and cannot be achieved through collaborative working;
- understand the synergy and trade-off between different types of benefits and costs associated with land and water management options;
- design and promote new forms of land and water management that can deliver intended outcomes more cost effectively; and,
- design targeted policies that reward land managers for providing the desired range of beneficial services.

Figure 10: [147]Graphic Representation of the Different Benefits Arising From the Beckingham Marsh Floodplain Ecosystem, on the River Trent, in Nottinghamshire,

A - 2006 land use compared with land use scenarios for maximum agricultural production and flood storage options

B - 2006 land use compared with land use scenarios for agri-environment and maximum biodiversity options

The graphs showing the changes in benefits with alternative floodplain land management scenarios to maximise different benefits. The benefits are classified by functions (P-production, such as agricultural production, R-regulation, such as drainage or water quality, H-habitats, such as maintenance and enhancement of biodiversity, C-carrier, such as infrastructure and buildings, and I-information, such as amenity and recreation benefits).

The Total Economic Valuation (TEV) of the flow of natural capital, based on direct and indirect use values (Box 16), will always be less than the Total Systems Value, which includes the infrastructure value associated with intact natural capital stock. The processes that directly give rise to ecosystem services are referred to as 'functions'. In the example of a wetland (Figure 11), one 'function' is slowing of water passage that provides the benefit of flood protection.

This valuation approach reflects the usefulness of the benefits, but value is also associated with scarcity, in that with an abundance of such benefits users will be less willing to pay for further benefits. In the case of wetlands, increased numbers in a particularly locality will reduce the biodiversity tourism benefits of individual wetlands. Where ecosystem service provision can be shown to be directly dependent on biodiversity (such as pollination services, Box 3), then the loss of biodiversity can be valued in terms of benefits foregone or reduced, but this requires a robust description of the relationship between biodiversity and ecosystem services (Section 2.2).

Box 16 Total Economic Valuation

The Total Economic Value (TEV) conceptual framework views ecosystem goods and services as the flows of benefits to humans provided by the stock of natural capital (Section 2.3). Values are assessed through the ways in which ecosystem services support people's own consumption (use values) and provide intangible human benefits (non-use values).

Use Values

Use values (figure 11A) can be further sub-divided into direct-use value and indirect-use value:

- Direct use values arise from direct human use of natural resource systems, including extractive use values from flows of benefits such as timber or fisheries and non-extractive use values such as tourism and recreation.
- Indirect use values result from the regulatory or supporting ecological processes that contribute to the ecosystem services giving rise to benefits. For example processes occurring in wetlands remove excess nutrients, improve water quality and provide flood protection through retaining water.

Non-use values

Non-use values (figure 11A) do not involve direct interaction between humans and ecosystems and include:

- altruistic values derived from knowing that others that others can enjoy the goods and services from ecosystems.
- bequest values are associated with knowing that ecosystems are passed on intact to future generations.
- existence values, arising from the knowledge the ecosystem and its services continue to exist.
- option value refers to the benefit from the security of knowing that ecosystem is being preserved for possible but unforeseen future uses, such as a species with a pharmaceutical applications.

Economic, deliberative and participatory methodologies are used to try to ascertain use values of ecosystem service benefits. These attempt to establish either individuals' willingness to pay (WTP) for an ecosystem service (or to avoid its degradation) or willingness to accept (WTA) compensation for its degradation (or for forgoing an improvement or restoration of an ecosystem service). Five main sets of methodologies are employed (see also Box 4, Environmental Accounts for Agriculture), which will be appropriate depending on the application and data available:

- Market prices, which can be used to estimate the value of ecosystem goods that are traded in formal markets, such as timber and fish. The prices need to be adjusted for any environmental market distortions.
- Cost methods, based on the cost of damage caused by the loss of an ecosystem service, or expenditure to prevent that damage, or the cost of replacing the ecosystem service altogether.
- Revealed preference methods, such as the travelling and access costs people are willing to pay to use an ecosystem for recreational purposes;
- Stated preference methods; such as surveys to determine people's willingness to pay for ecosystem services in hypothetical markets; and,
- Deliberative and participatory valuation methods ranging from group-based deliberative monetary valuation to citizens' juries.

Values for the many ecosystem services that are not directly traded in markets must be derived through the last four sets of approaches. These often require extensive time, skills and data, and the findings are sometimes disputed. However, as the number of robust primary valuation studies of ecosystem services grows, it is feasible to transfer these estimates to assess values in other situations. New insights on how humans value items beyond their market price are arising from disciplines such psychology and neuroscience in combination with behavioural economics. These approaches are seeking to provide a better understanding of how values are constructed and how they change in response to changing circumstances and external factors, such as perceptions of threat or opportunity, how they vary with knowledge and experience, and how they are constructed by individuals or groups working collectively.[149]

4.7 Using Valuations in Policy Making

Valuation can be used to:

- understand the contribution that an ecosystem makes to an area
- determine whether a policy intervention is worthwhile
- the costs and benefits for different stakeholders of an ecosystem or proposed changes to it
- identify financing sources to maintain ecosystem services.[150]

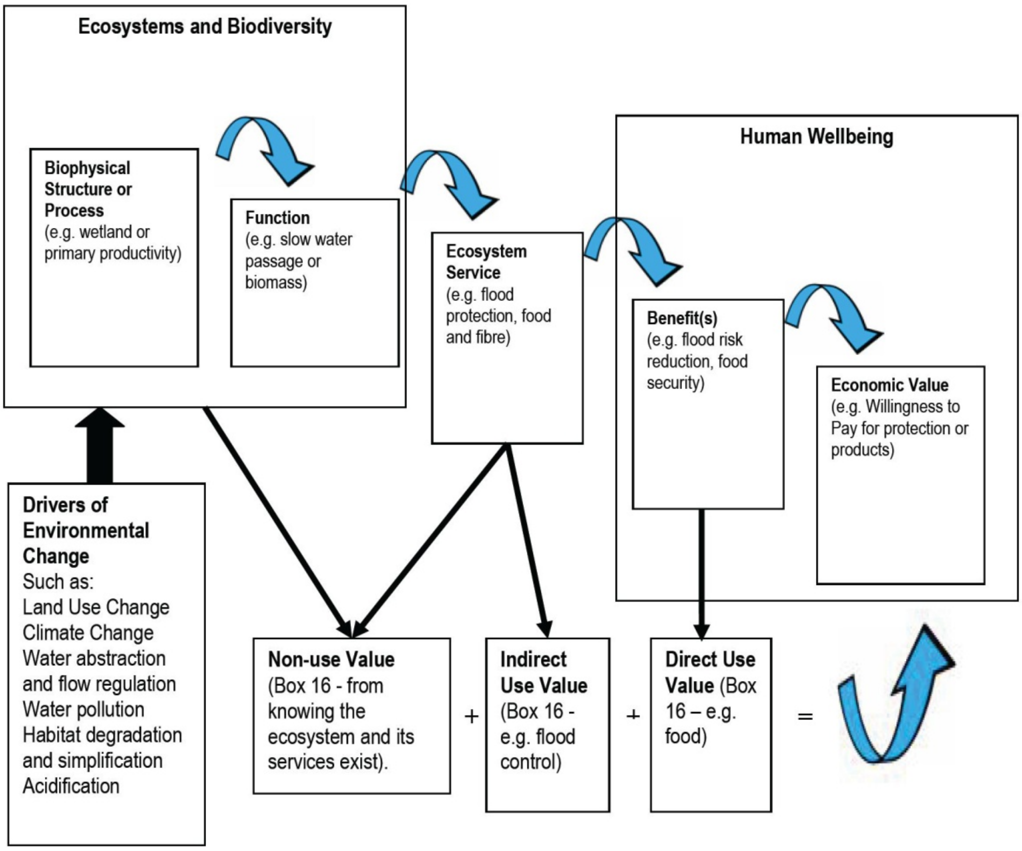
There is an extensive literature on the various valuation methods, the contexts in which they are applied and their limitations.[151] It is neither practical nor necessary to produce an economic valuation study for every policy decision and value transfer methods have been developing to make the best use of existing valuation studies within decision making (Box 9). The complexity of the attitudes, preferences and values that people hold can pose problems in applying valuation techniques to decision-making and the use of some methods remains contentious.

The approach of using willingness to pay surveys of individuals to establish valuations to inform decision making (Box 16), has been challenged by research that suggests that people would rather collectively take part in decision making in forums where they can deliberate on issues rather than place sole reliance on valuation.[152] A recent review has concluded that there is scope for better guidance on the selection, design and application of the different methods, and a need to include tests for rigour and robustness of the analysis and results.[3]

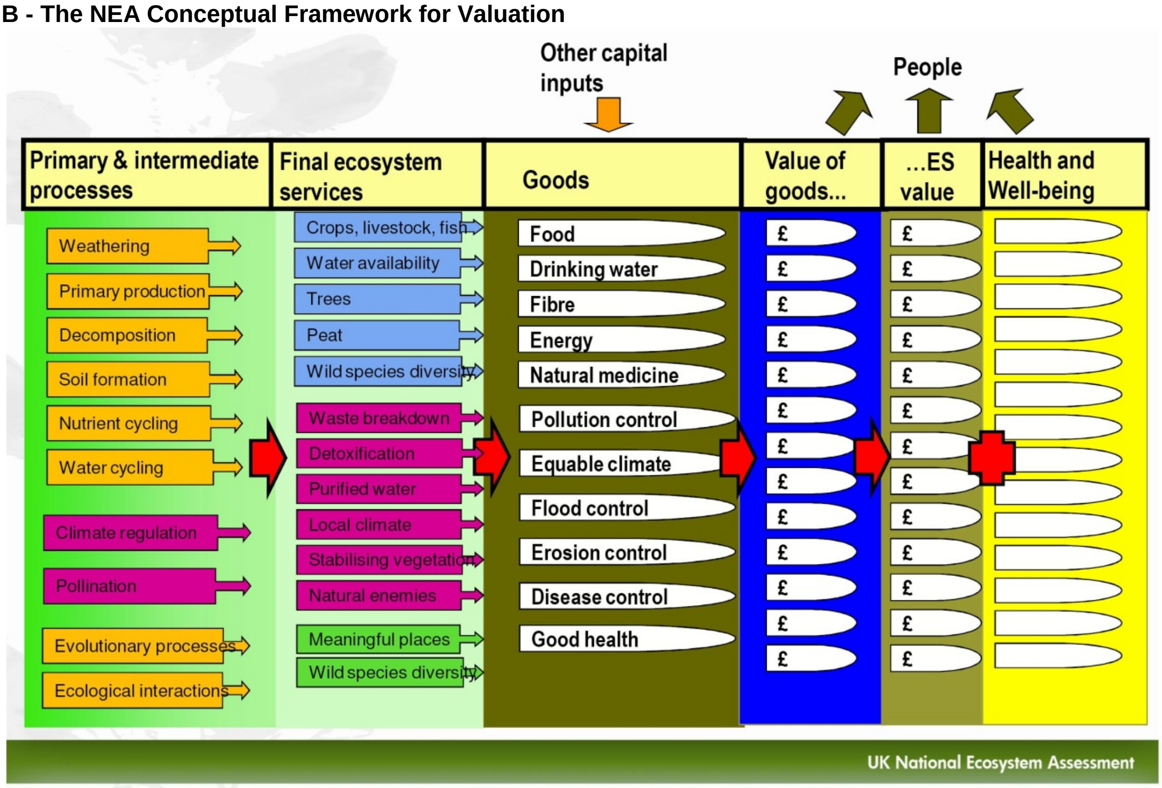
Valuation also varies significantly with how aware people are of background issues and with their socio-economic status, and it is widely recognised that valuation of ecosystem services is highly context specific and should be guided by the perspectives and requirements of beneficiaries. There is a need to distinguish between benefits and values clearly, because different groups may have different values on benefits.

Figure 11

A-The Relationship of Ecological Functions to the Ecosystem Valuation Process[153]



B - The NEA Conceptual Framework for Valuation



The Framework differentiates between the final services that directly deliver goods and services to people from the intermediate processes and services upon which these depend

While the capacity of ecosystems to deliver benefits to people may be constant, the values attached to them can change over time.[85] However, this subjectivity and ephemerality could conflict with the 'impartial nature' of CBA and governance structures acting in the national interest.

Critics of valuation question whether the true future consequences of the loss of any natural capital asset can be anticipated with any confidence. In addition, the difficulties in making a spatially explicit quantification of benefits and costs are cited, which identifies the suppliers and users of ecosystem service benefits to design effective and equitable policy interventions. For example, the SCaMP project (Box 14) allows a water company to make payments to farmers to help to reduce the impact of agriculture on water resources. The valuation of SCaMP's ecosystem service benefits is incomplete due to information gaps and the long term nature of the benefits, but appears to outweigh the economic costs.

The initial proposals for SCaMP were not supported by the regulator OFWAT, on the grounds that the costs arising from the environmental impacts of agriculture should not be borne by the water consumers (POSTnote 320). This was further complicated because the water company bearing the costs of the impacts was also the farmers' landlord. However, the project was endorsed by the Defra Secretary of State in recognition of both the economic value of the benefits arising from environmental measures and the research value of implementing innovative land use management measures.[154]

Where possible, valuation should be spatially and temporally explicit at scales meaningful for policy formation or interventions, as the ecological processes giving rise to benefits are specific to areas and occur over differing time scales. For example, a cost benefit analysis of multi-purpose woodland compared with retaining land in agricultural grassland in Wales, using ecosystem service valuation, has shown that that an increase in woodland cover, substituting for sheep grazing, would be cost beneficial in many parts of Wales. The analysis also showed that existing forests are not optimally located to fulfil their potential; the ideal places would be adjacent to population centres where recreational benefits are highest.[155]

4.8 Institutional Frameworks

For ecosystem service provision to be maintained within environmental limits, management decisions should keep future options open in spite of uncertainties. For example, the loss of unique aspects of natural capital, such as the tidal bore in the Severn Estuary (Annex D), could breach environmental limits as it shuts off future options for the provision of ecosystem services. However, the kind of governance structures and institutions most capable of delivering the ecosystem approach, maintaining the required levels of natural capital and sustaining the flow of ecosystems in the long term remains.

The European Academies Science Advisory Council (EASAC) has recommended that one means of maintaining delivery of ecosystem services would be an EU Ecosystem Services Directive, analogous to the existing EU Habitat Directive that sets out the strategy and targets for biodiversity conservation in Europe.[156] It suggests that an Ecosystem Service Directive would set out a strategy for the conservation and maintenance of ecosystem functions (Section 3.2) and protection of the levels of ecosystem service benefit provision for not only European populations but also globally.

The EASAC report suggests that an Ecosystem Service Directive could establish priorities with two technical annexes setting out the key ecosystems of Community interest and the service providing units of Community Interest. As described in Section 2.4, service providing units are populations of species critical to the provision of particular ecosystem services.[157] This can be broadened through the parallel concept of ecosystem service providers, which are ecosystem wide or community attributes characterised by the component populations, species, functional groups or habitat types that collectively provide ecosystem service benefits.[158]

Impact Assessment Methodologies

To maintain natural resource systems within environmental limits, the appraisal of government policies and projects should ensure that the value of natural capital and ecosystem services is considered and future costs arising from any increase in environmental risks are identified. HM Treasury's 'Green Book', *Appraisal and Evaluation in Central Government*, provides overarching guidance on how to appraise and evaluate government intervention, setting out the main stages in the appraisal process. For any new proposal, "…the effects on the environment should be considered, including air and water quality, land use, noise pollution, and waste production, recycling and disposal".

Several methodologies have been developed to take account of the environmental impacts of policy decisions, land use changes and other pressures on natural resource systems, including:

- Environmental Impact Assessment (EIA) carried out for individual projects (Annex D).
- Strategic Environmental Assessment (SEA) of plans and programmes (Annex D).
- Regulatory Impact Assessment (RIA) or Impact Assessment, the policy assessment frameworks used to examine and measure the likely benefits of new or changed policies and regulations (Annex D).
- Cost-Effectiveness Analysis, which compares the relative costs and outcomes (effects) of two or more courses of action (as opposed to Cost Benefit Analysis which assigns a monetary value to the effect of a decision).
- Life-Cycle Analysis (LCA), which compares the full range of environmental and social damages assignable to products and services, to be able to choose the least burdensome one (Annex B).
- Environmental Risk Assessment, which is conducted to evaluate the likelihood that adverse ecological effects could result from a given course of action or decision (Chapter 6).

There is substantial academic literature on the strengths and weaknesses of these various approaches, some which are required by law. A brief summary of the existing relevant aspects of the regulatory frameworks for assessing proposed plans, programmes (SEAs) or projects (EIAs), as opposed to policies, is set out in Annex D. The recent Government Economic Service "Review of the Economics of Sustainable Development" suggested that the existing high level principles of UK Sustainable Development Strategy are too general to indicate whether a policy is consistent with the principles or how to make trade off when they conflict.[159]

Globe International's "Natural Capital Action Plan" recommended that all policy and project proposals that influence the environment should undergo an economic appraisal that includes the valuation of ecosystem services, with government departments obliged to incorporate a costed explanation of how their policies will enhance or deplete natural capital/ecosystem services.[160] If environmental limits have been defined, such costings could be used to determine whether policies are increasing the risk of exceeding limits or decreasing the likelihood of this occurring (Section 3.5).

Applicability of CBA

Economic analysis, in the form of Cost Benefit Analysis (CBA) is the most frequently used policy decision support tool used to quantify trade-offs between economic benefits and environmental and social losses. Other approaches can be both complementary and provide inputs for, or they can act as alternatives to, CBA. CBA is used to set out the relative merits of alternative policy approaches when advising Ministers in the UK, in accordance with the principles set out by the HM Treasury. It is beyond the scope of this report to discuss in detail whether consideration of ecosystem service provision and environmental limits could be most effectively implemented through CBA, other approaches or combination of approaches.

The role of economic analysis in environmental policy is to determine where a change in practices or policies may be in the wider public interest. CBA was developed to assess the impact of marginal changes, such as the costs of small changes in air quality as a result of a policy decision to expand an airport. Although this decision may be considered insignificant when assessed in isolation, it may be significant when evaluated in the context of the combined effect of all past, present, and reasonably foreseeable future activities that may have or have had an impact on air quality.

CBA is generally regarded as being best suited to informing the most appropriate means of mitigating a risk rather than if a risk should be managed (Chapter 6). Decision choices, particularly 'downstream choices' about project or policy design or management plans, need a wider array of information than just valuation, such as participatory methods (Section 4.9). No single approach, such as CBA, is likely to suffice and a whole toolbox of approaches will be needed to deal with ecosystem based management issues.

A recent example of values for ecosystem services being used in an impact assessment was for the 2009 Marine and Coastal Access Act, which used the economic value of marine ecosystems to inform the CBA of the impact of proposed measures, such as marine spatial planning and marine conservation zones contained in the Act (Table 4).

Incorporating Ecosystem Services

Information on ecosystem services at scales useful for determining how policy or land use choices will affect human wellbeing is necessary for effective assessments of impacts. These should incorporate understanding of linkages between changes made and their knock-on impacts. This would include the need for compensatory or mitigation measures to offset the consequences of changes to ecosystems and ecosystem processes and benefits from the local to the national level.

For example, urban development proposals do not generally consider impacts on soils or of soil sealing on water infiltration, with knock-on consequences for drainage and drainage systems. Information on levels of ecosystem services and subsequent economic analysis can indicate the most cost-effective means of achieving required levels of benefit delivery. It can also be used to assess changes in the distribution of costs and benefits across stakeholder groups as a consequence of a decision or policy.

To inform the development of effective policy frameworks there would need to be consideration of:

- Explicit measures of the condition and trends of biodiversity associated with the relevant ecosystems.
- Determination of the delivery of ecosystem benefits in biophysical terms (both quantity and quality), to inform economic valuation or measurements required. This should include quantification of how different types of uses of ecosystems change the delivery of benefits.
- The context of contrasting future scenarios, which incorporate both the value of ecosystem services and the cost of actions affecting those ecosystems, so that the impacts of alternative decisions on ecosystem services can be assessed.
- Integration of an analysis of risks and uncertainties, including the limitations of knowledge of the impacts of human actions on ecosystems.
- Economic valuation applied to changes in services, which requires a good understanding of the service flows and the determinants of demand.
- Understanding of the role of property rights and entitlements to the use of ecosystem benefits (Section 5.3).

The NEA (section 4.3) includes the development of scenarios to inform ecosystem assessments. These provide an understanding of the current state and past trends in ecosystem services and the likely policy or management responses that might be appropriate given a range of possible futures. [136] Scenarios are not predictions about the future, but are a set of tools to explore issues that involve high uncertainty and high complexity, such as environmental limits.

Decision-makers, such as parliamentarians, will want to be aware that there are ranges of potential outcomes for any decision, and so that they can judge the capacity of proposals to withstand uncertain future risks (Chapter 6). Scenarios can be used by policymakers or stakeholders to:

- consider possible long term consequences of decisions
- assess the implications of future uncertainties for various management options; and,
- enhance stakeholder participation (Section 4.9) by representing conflicting opinions and different world views.

4.9 Trading-off Gains and Losses

The institutions and systems that govern use of natural resource systems are a common source of societal tension, especially when the different values attached to aspects of natural capital are traded off against each other. Policy decisions and changes in use of natural resource systems lead to actions that impact on ecosystems, causing changes in ecosystem structure and function and ultimately in benefits.

The 'right' decision in economic analysis has a precise meaning: a decision that, on the whole, has more benefits to the society than costs.[154] All ecosystems deliver a broad range of services (listed in Figure 3), for some of which biodiversity is crucial and some of which have particular economic or social value.

Table 4: Values of Goods and Services Delivered by UK Marine Biodiversity[161]

Ecosystem Service Category	Ecosystem Good/Service	Monetary Value
Provisioning Services	Food provision	£513 million
	Raw materials	£81.5 million
Regulating Services	Gas and climate regulation	£420 million - £8.47 billion
	Disturbance prevention	£ 17 - 32 billion
	Bioremediation of waste	No data
Cultural Services	Cultural heritage and identity	No data
	Cognitive values	£317 million (2002)
	Leisure and recreation	£11.77 billion (2002)
	Non-use values	£0.5 – £1.1 billion
Supporting Services	Nutrient cycling	£800 – 2,320 billion
	Resilience and resistance (See Section 5.2)	No data
	Biologically mediated habitat	No data
	Option use value	No data

For example, a forest can:

- be a major store of carbon, helping to regulate climate
- be a resource for industry in the form of fibre or fuel
- prevent loss of soil and nutrients, flooding and avalanches
- play a role in the water cycle, ensuring cycling of water vapour back to the atmosphere
- provide a location for recreational activities.

The concept of valuation is associated with trade-offs, as a given benefit has the value only of what humans are willing to pay or forgo for that benefit. Benefits arising from ecosystem services will be valued differently and this information can be used in cost benefit analysis to quantify the trade-offs involved in policy and planning decisions.

However, as the distribution of costs and benefits is likely to be unequally shared, cost benefit analysis (CBA), normally assumes potential compensation. Those that benefit from changes are assumed to be in a position to compensate losers. However, this rarely occurs in practice, particularly in the case of changes in the use of natural resource systems. Actual monetary compensation to the losers in the delivery of ecosystem service benefits from changes in use or management of an ecosystem, if used with relevant ecosystem service benefit valuation criteria, may make it easier to ensure that the most viable policy option is pursued.

Compensation could also be provided in the form of the recreation or restoration of habitat elsewhere to deliver ecosystem services. This is already required by the EU Birds and Habitats Directives to mitigate for the loss of areas designated a Special Area for Conservation (SAC) or Special Protected Area (SPA). These requirements could be extended to areas that currently do not have statutory protection, a process usually referred to as biodiversity offsetting (POSTnote 369), to ensure no net loss of ecosystem services (Section 5.7).

Not all sites will deliver the same level of provision of ecosystem service delivery. For example, under current ecosystem service valuation frameworks, sites near high levels of population density, such as the Barnes Wetland Centre in London, may provide higher levels of cultural benefits than a remote site, such as in the highlands of Scotland. Other habitats, such as ancient woodland or chalk grassland are so rare, unique and locally specific that valuation is not applicable and recreation of compensatory habitat is not possible within relevant time scales.

Where site loss is compensated for through the creation of alternative sites, the provision of ecosystem services should be a key consideration to ensure that relevant human users still have the same levels of benefits provided. Although the relevant methodologies have yet to be developed, comprehensive offsetting may be critical in ensuring that levels of ecosystem services are maintained within environmental limits in a given locality.

Acknowledging Loss of Benefits

Ensuring that any loss of benefits is acknowledged within decision-making or management systems, by attaching an appropriate value to trade off against economic gains, increases the transparency of decision-making. If an ecosystem is primarily managed to deliver one ecosystem service, this will reduce levels of other ecosystem services supported by the ecosystem. For example, a forest managed exclusively for timber production, may have less recreational value, store less carbon and be less effective at retaining nutrients. A key role for scientific advice is to provide understanding of these relationships between services and how best to manage their interaction.

The Foresight study on land use has called for decision-making frameworks that encourage multifunctional land use to create landscapes that are more “resilient” in the long term. It highlighted the need for local planning authorities to consult formally with local residents on options, benefits and trade-offs for new forms of development. Many ecosystem services are either undervalued or have no value in current decision making frameworks, although crucial to human well-being.

Benefits over a long-term horizon from regulating and supporting ecosystem services, such as climate regulation and flood alleviation, are frequently ignored. Decisions take more account of shorter term private gains in benefits (such as increased agricultural productivity from wetland drainage) than longer term loss of public benefits (such as increased risk of flooding and decreased water quality). The benefits generated by ecosystem goods and services are both private and public goods and occur over a range of temporal and spatial scales and can be associated with a variety of property rights and other institutional arrangements.

The gainers and losers from any environmental change vary depending on the type and scale of ecosystem service provided, the mix of stakeholders involved, the economic characteristics and the cultural context. However, even if ecosystem services have no formal economic valuation, they can still be traded. For example, where conflicts arise in ecosystem service provision and management, 'compensation in kind' could be provided to users who have to make sacrifices to benefits to others, such as land swaps (a swap of land with the same value and size).

Stakeholder and Public Participation

A key tenet of the ecosystem approach is that it should involve all stakeholders and balance the wider public interest with local interests. The CBD stresses that the management of ecosystems and their services is a matter of 'societal choice',[162] and that individuals have a legitimate right to influence and shape decisions that impact on livelihoods and well-being (Annex C). This requires engagement with a broad range of institutions, organisations, groups and individuals that have an interest in, understanding of, or potential influence over, the management of a given problem.

The closer management is to the ecosystem, the greater the responsibility, ownership, accountability, participation, and use, of local knowledge. At the local scale decision making can accommodate consideration of multiple benefits, trade-offs between ecosystem service benefits, environmental limits and appropriate levels of stakeholder participation is possible. However, economic and participatory methods are complementary to existing decision making processes rather than alternatives. They provide new inputs into the process and facilitate debate and scrutiny of the reasoning and assumptions behind decisions.

Participatory approaches alone cannot resolve fundamental conflicts in the management of natural resources and their services. However, they can make clear the contested nature of many environmental decisions. Decisions are contested for a range of reasons including: [163]

- The underlying purposes of decision making are often open to different perspectives. Potential options for management can consequently be varied and in many cases contradictory.
- Decisions will be made in circumstances where evidence is often highly uncertain or incomplete.
- Concern for the management of ecosystem services must be compared with a range of other benefits driving and informing decisions.
- Management is complex, hence efforts to deal with issues in one area often simply expose difficulties elsewhere.

There is a substantial academic literature on the uses and challenges associated with stakeholder participation in natural resource management, which is beyond the scope of this report to summarise.[164] However, it is clear that participatory approaches in themselves are insufficient to ensure that environmental limits are not breached, given the complex set of values involved and the conflicts which can arise when ecosystem service benefit trade-offs are made, especially between local, regional and national scales.

Using Ecosystem Service Assessments

A recent study for the Government Office for the East of England, Natural England, Forestry Commission, Environment Agency and other bodies, "Valuing Ecosystem Services in the East of England", undertook ecosystem valuation case studies to show how the ecosystem services approach could be implemented in mainstream planning and governance processes at national, regional and local level.

The case studies were in five areas, Marston Vale, Cambridgeshire Fens, Blackwater Estuary, Norwich and Great Yarmouth, and sought to demonstrate how the approach can work in practice in a range of situations and to which policy and funding decisions it should be applied. The study found that ecosystem services were seen to be applicable at a range of scales from site-specific decision-making to local, sub-regional, regional and national levels, but that buy-in by stakeholders is critical and that targeted engagement is required.

Rather than using the full ecosystem service assessment framework, it found that the approach should be tailored to the situation and budget available and that a qualitative or a partial semi-quantitative assessment of the benefits arising from a site may be the optimal means of engaging stakeholders (Figure 12). The study also found that there was a paucity of relevant economic data for valuation of ecosystem service benefit in site-specific situations. The ecosystem service assessments linked well to existing policy tools such as sustainability appraisals, strategic environmental assessments, environmental impact assessment, cost benefit analysis and risk assessments.

There were concerns among stakeholders that a strict framework implemented from above would be a significant regulatory burden. However, consideration of ecosystem services in decision making was impeded by the lack of a national standard, accepted, transparent, robust, simple, clear and holistic ecosystem service assessment methodology.

In particular, it seemed that Environmental Impact Assessment might be a better informed process if it took account of the ecosystem approach (Annex D). There is often a tension between economic, social and environmental assessment in project appraisal. Ecosystem service assessment would clarify both the public benefits of the environmental changes and which stakeholders would bear the environmental costs. Ecosystem service assessment could also provide a basis for compensating the communities who bear environmental costs at the local scale for public benefits at larger scales.

A key consideration is whether the existing EIA framework is sufficiently coherent and comprehensive to represent the diversity of service flows (as well as the impact on stocks of environmental assets). Further studies are required to determine whether the ecosystem approach could be practically relevant and operational at the level of decision making, or whether it would make the process more unwieldy and complicated.

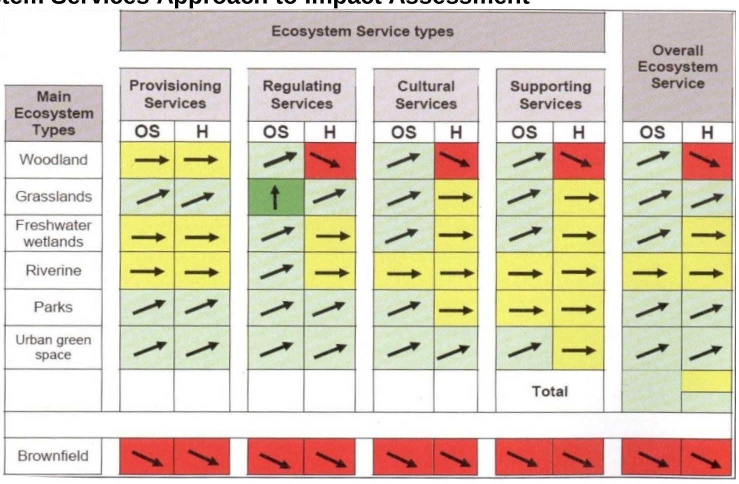
One challenge to engaging stakeholders is that the language of 'ecosystem services' and 'the ecosystem approach' may be a 'jargon' barrier to engaging stakeholders. Studies have found that people misunderstand a range of terms associated with ecosystems services, including ecosystems, biodiversity, green infrastructure, provisioning, regulating, cultural and supporting services. By contrast, terms such as 'natural resources' and 'benefits from nature' are readily understood. [165]

Participation, Assessments and Limits
Defra has recently consulted on guidelines on 'participation and an ecosystems approach to decision making'.[163] The draft guidance suggests that participatory and deliberative techniques (PDT) could be incorporated into policy appraisal and evaluation procedures in accordance with the HM Treasury 'Green Book' (Annex D). Participation should start at an early stage to both clarify the issues to be addressed by the decision making process and where the priorities of stakeholders lie and to identify constraining factors.

Participation should also be informed by the environmental baseline to identify and categorise ecosystems and their services impacted by policy options (Section 4.6), against which changes in service provision can be assessed and valued as well as identifying which stakeholders will benefit or lose (Figure 13). [163] However, if the policy question under consideration is not primarily environmental, changes in ecosystem services may become apparent only during formal appraisal procedures (Annex D), requiring decision makers to 'work backwards' to identify affected stakeholders (Section 4.11). Formal appraisal includes consideration of non- monetary and monetary costs and benefits (Section 4.9), including cost benefit analysis, although other techniques may also be required. [163]

There is also a range of techniques for stakeholder engagement, depending on its objective, be it to inform stakeholders, to learn about stakeholders or to work with stakeholders. Only the last of these is relevant to environmental limits, in particular deliberative approaches that actively and explicitly involve stakeholders in decision-making. For example, the use of deliberative multi-criteria analysis to judge the non-monetary and monetary costs of different management options. [163]

Figure 12 An Ecosystem Services Approach to Impact Assessment [166]



The example shown from the East of England study compares service impacts of land conversion to Housing or open space (OS). The arrows show trends in services and shading shows levels of significance of impact, red a significant negative trend and green a positive trend.

Defining Stakeholders
The term 'stakeholder' is an umbrella term for organisations, groups or individuals affected by an issue and is there is no accepted standardised way of categorising relevant participants. The Defra guidance suggests grouping stakeholders in three ways: [163]

- Based on competencies, such as scientific, technical or procedural competencies.
- Based on power, identifying stakeholders that can influence the success of the process through either designated formal authority, power of resources such as property rights or network power in relation to the decision making network.
- Based on impacts, especially those that directly or indirectly bear costs.

Stakeholders can also be grouped according to ecosystem service outputs, such that farmers are competency-based 'provisioning' stakeholders, whereas recreational users of farmland are directly impacted 'cultural' stakeholders. These types of techniques can be used to characterise stakeholders for environmental decision making, such as determining environmental limits for a given natural resource.[167]

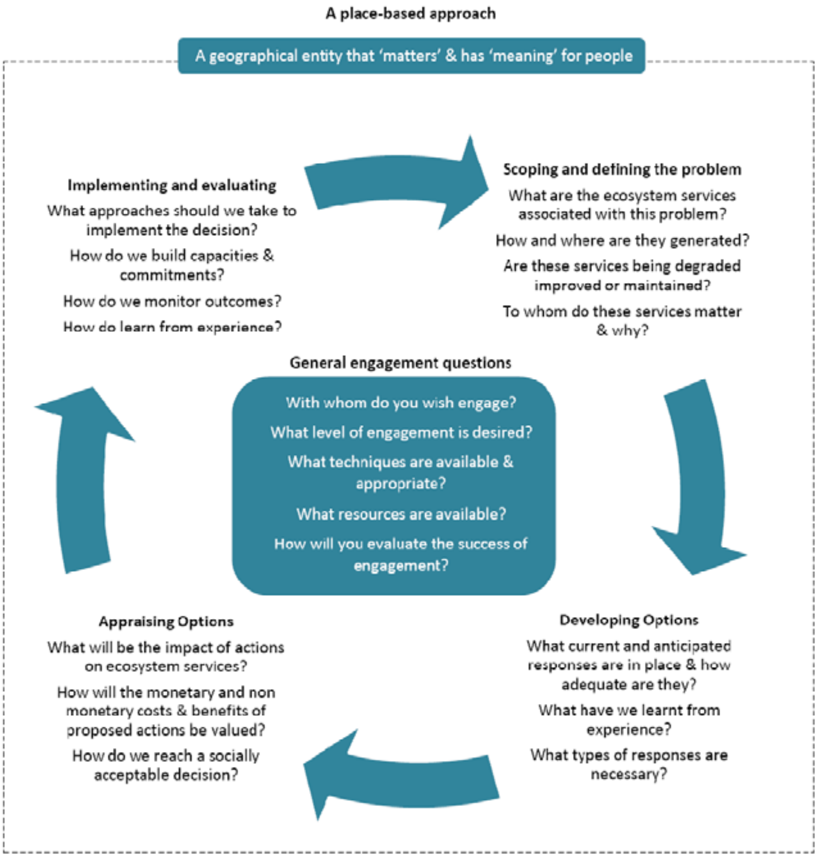
Water Framework Directive

One policy area where the status of ecosystems needs to be considered arises from compliance with the requirements of the Water Framework Directive (WFD) to achieve 'good ecological status' in freshwater bodies (POSTnote 320). The Directive requires member states to:

- classify "water bodies";
- determine the ecological status of these water bodies;
- determine 'pressures' on water bodies that are not in 'good ecological status'; and,
- to put in place cost effective 'measures' to bring them in to good status within set timelines.

The Water Framework Directive encourages management at river catchment scales to protect inland and coastal waters, as well as groundwater. Current statistics suggest that only 22% of rivers and 25% of all water bodies met the WFD's "good ecological status" target in 2009 and this figure is likely to increase only to 30% by 2015. In 2009, the Environment Agency published 'River Basin Management Plans' (RBMPs, Postnote 320), that classify and set objectives for water bodies in England and Wales.

Figure 13 The Decision Cycle and Participation in an Ecosystems Approach: key issues[163]



These plans aim to take a "whole systems" approach to managing the pressures on catchment systems in conjunction with the Flood and Coastal Erosion management. Management strategies are being drafted, on which stakeholders are consulted, that may restore natural processes in river catchments and coastal waters, with a subsequent increase in the benefits arising from these ecosystems.

A critical aspect of achieving this will be land and water management measures, as the way in which land is used and managed impacts water quantity and quality (such as fertiliser application to agricultural land). The implementation of the WFD in the UK has been criticised for falling short of systemic integration of land and water management at the river catchment scale.[168]

Achieving such large scale changes in management of natural resource systems requires engagement of stakeholders and the identification of users and beneficiaries. Stakeholder uses and interests and the influences they can deliver in terms of effective management measures of pressures on 'water bodies' are a key aspect, as well as the need to balance competing local priorities for ecosystem service benefits (Box 17).

4.10 Accommodating Uncertainty

Economic valuation of biodiversity offers ways to compare the benefits and costs associated with changing ecosystems within decision making frameworks, but it does not capture information about non-economic values or all the dimensions of human well-being, such as many cultural services. Although the economic values of changes in ecosystem benefits are important for decision-making they do not replace the need for political deliberation.

The Organisation for Economic Co-operation and Development (OECD) has recommended that decision-making processes should take account of all influencing factors.[169] There will always be uncertainties, whether physical, ecological or economic, as well as the changes to which monetary values cannot be ascribed. In short, moral obligations, intrinsic values and other ethical issues need to be taken into account, alongside cost-benefit analysis, in trade-offs. However, the degree to which factors that cannot be fully monetised, such as moral and ethical environmental issues, influence the development of the business case for policies at the UK Government Department level is not clear.

Decision-making frameworks to allow trade-offs require ecosystem service indicators to allow progress towards relevant targets to be assessed (Annex B). Where there are scientific uncertainties in relation to the management decisions, these can be addressed through monitoring the outcomes of land use change to provide further understanding of how any given ecosystem functions. The decision making processes can then be adapted accordingly (Section 5.4).

Discounting and Environmental Limits

Discounting is used in Cost Benefit Analysis to compare economic impacts occurring at different times. It is generally assumed that consumption in the future is worth less than consumption now, and so a discount is applied to the future value of natural capital benefits. This assumption is made on the basis that:

- Humans have an innate preference for consumption in the present rather than the future.
- Consumption levels in the future are expected to be higher so individual units of natural capital benefits will be worth less.
- Improved technologies will replace the services delivered by ecosystems and future consumption levels are uncertain – so that the need for natural capital benefits by future generations cannot be assumed.
- Economies will continue to grow at a rate of 3% each year and future generations will have greater wealth to deal with the environmental problems caused by current economic use of natural resources.

There is a substantial literature relating to intergenerational equity and the validity of discounting, and this issue was discussed at some length in the 2006 Stern Review on the Economics of Climate Change. The Stern Review argued for a low discount rate, thereby substantially increasing the likely future costs of climate change impacts compared with previous estimates using different discount rates. Increasing future costs makes the mitigation of emissions now more cost effective than attempting to rectify the future impacts.

In general, a higher discount rate will lead to the long-term degradation of biodiversity and loss of ecosystem services. A 5% discount rate implies that loss of natural capital 50 years from now will be valued at only one seventh of the same amount of biodiversity loss today. The TEEB report has argued that the present rates of degradation of natural resource systems are not suited to standard economic approaches and should not be subject to standard discount rates. Several leading economists have also suggested that the standard economic model offers inadequate framework to analyse environmental issues characterised by irreversibility, uncertainty and long time horizons.[170] How the current levels of use of natural resource systems will affect future generations remains difficult to discern, given the uncertainties about where ecological thresholds lie, and the extent to which these changes will affect future human wellbeing. Discount rates could be informed by an assessment of the kinds and quantities of natural capital assets that are expected to be passed on to future generations and how these would be affected by policy options.

Box 17. The Norfolk and Suffolk Broads

The Broads comprise a complex (freshwater, brackish and saline zonal area) wetland located in the East Anglian region of England. It is the largest protected wetland and third largest inland water body in England. The Broads have the status of a National Park and are managed by an official agency, the Broads Authority (BA), with powers similar to other UK National Park Authorities. They are the most visited lowland wetland in England. The rivers and connected broads (shallow lakes) are intensively used for recreational boating, involving around one hundred boatyard operators. The direct causes of wetland degradation/loss are:

- Land use change and agricultural development, the impact of agricultural activities over the years has changed from drainage and land use effects to diffuse nitrate and phosphate pollution that results in an excess of nutrients in water bodies (eutrophication).
- Further growth of water-based tourism, with the number and popularity of motor boats causing congestion and noise pollution at various locations in the systems
- Increased risk of saline intrusion and flooding, the east coast of England is at risk from saline inundation from the North Sea and the Broads area is also prone to fluvial (river) flooding. There are programmes to improve and maintain flood defences, but with rising sea levels, saline intrusion will continue regardless of an improvement in sea defences.
- Neglect of fen and carr woodland habitats (woodlands that develop on unmanaged fen habitat).

The East Anglian Region is expected to accommodate a major increase in housing, which will have a significant impact in terms of water resource use and the amount available to improve wetland ecosystems. Surface water throughout the region is already fully committed to existing abstractions and the environment during the summer and there are no significant quantities of water available as backup although winter surface water is still available in most of the region. Any lowering of the water table would have a significant impact on the fenland ecosystems. There is unlikely to be sufficient resource to protect all of the Broads to a uniform standard. To respond to climate change impacts, there will need to be a more flexible catchment-based approach that uses natural processes to accommodate change, such as using washlands to store floodwaters during peak flooding events.

The core purpose of the BA's statutory duties is focused on the requirements to balance navigation, biodiversity conservation and recreation/amenity interests. It has to operate by sometimes making pragmatic trade-offs, subject to EU Directives and national legislation constraints. Current strategy focuses on improving areas least likely to suffer future saline intrusion or in inundation, while preventing any further degradation of the wetlands as far as is practically possible. The BA and other official agencies in the broads have to take account of the effects of environmental change to restore and improve water quality and have adopted a restoration strategy which seeks to undertake waterbody management within a more naturally functioning flood plain over a time horizon of 50 to 80 years. The BA also seeks compliance with existing legislation such as the Water Framework Directive and the achievement of "good ecological status". However, some of the regulatory approaches to biodiversity conservation set out in the EU Birds and Habitats Directive have hindered the process of trading-off political, economic and environmental priorities.

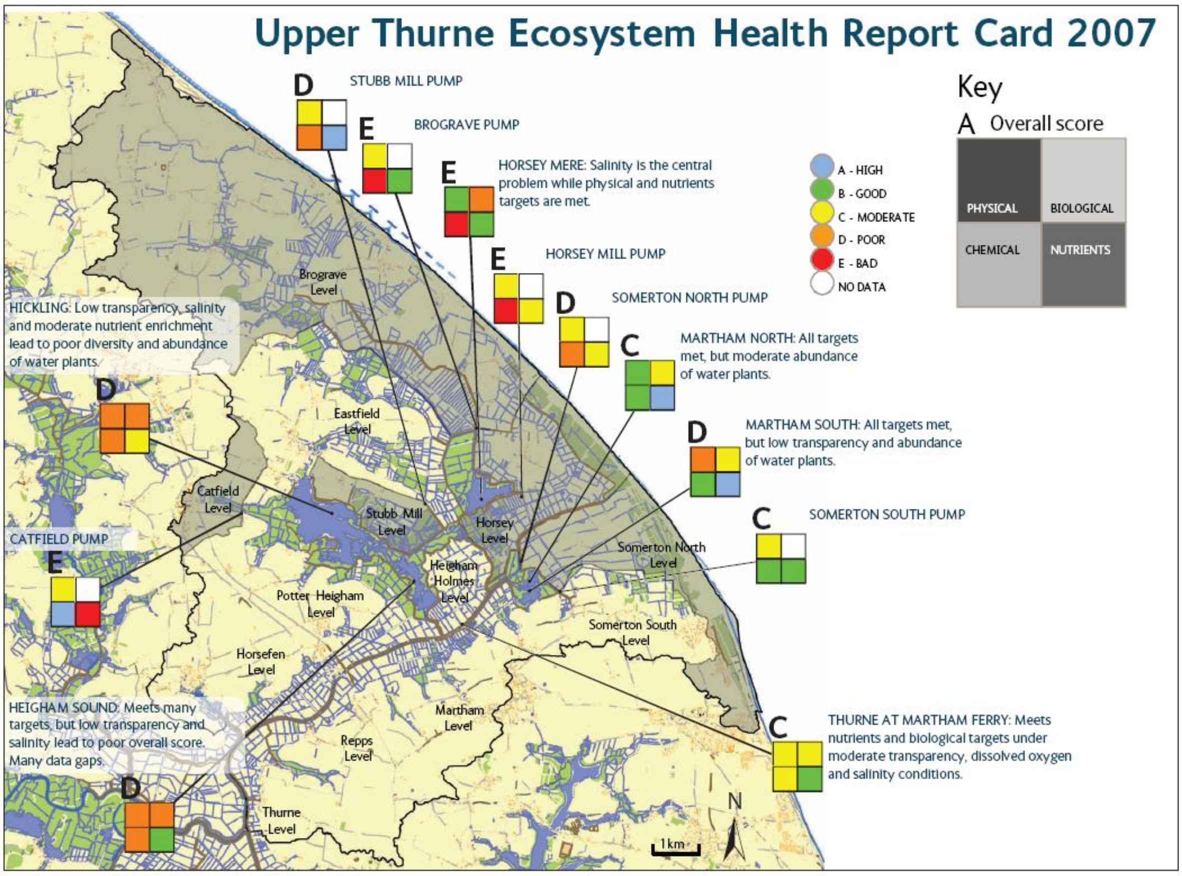
Regulatory approaches are also constrained by insufficient scientific data and the mismatch between environmental processes and administrative boundaries. For example, the improved water quality on Hickling Broad accelerated aquatic plant growth, including rare species covered by the Habitats Directive, to the extent that navigation of non-diesel powered craft was impeded. This created a clash between the statutory duty to maintain navigation access as well as with wider environmental concerns to discourage use of diesel powered craft and the need to maintain the boating industry. A dialogue with a range of stakeholders and government agencies led to a compromise plant-cutting solution, and highlighted the need for more stakeholder inclusion and a flexible approach to interpreting EU Directives. The BA has since re-organised itself and has set up a stakeholder forum to input into the main committee of local elected politicians and members selected by the government to represent national interests. There is an intention to include local parish council members on the main committee, but there may be tensions between managing at strategic catchment scale and local concerns.

A recent Rural Economy and Land Use Programme (RELU) research project[171] has demonstrated how stakeholders can be engaged at a catchment level to facilitate a collaborative and adaptive approach to catchment management to protect rural land and water resources using an 'ecosystems health report card' on the River Thurne in the Broads. This is similar to approaches used in Australia. The report card (see Figure 13) is based on the calculation of an "ecosystem health score" that compares the measured value obtained for a water sample against both an agreed target and a worst-case scenario. Parameter values can include physical, chemical, nutrient and biological information for a chosen sampling site that represents a water body in a catchment or sub-catchment. The scores on the card can be compared with the EU Water Framework Directive ecological classes, although the WFD classifies waterbodies on 'one out-all out' basis, whereas the report card seeks to report incremental improvements and highlight where specific mitigation measures could be targeted. Although they are based on the best available information, scientific uncertainties mean that targets and worst case scenarios cannot be set for all relevant categories, such as suspended solids, but the system is designed to incorporate changes in targets when new knowledge becomes available. The report card is used to:

- provide an easy to understand snapshot of the health of a catchment's freshwater and estuarine/marine environments in relation to environmental targets and standards;
- raise awareness of change in the condition of waterways over time;
- build understanding of the effectiveness of improvements in land and water management;
- focus management efforts and resource allocation to protect environmental assets identified by the community; and,
- demonstrate possible future scenarios.

Figure 14: The Ecosystem Health Report Card (Box 17) for the River Thurne Catchment in the Norfolk Broads.[172]

A-Report Card Front Page



B-Report Card Back Page

Indicator	Parameter	Martham Ferry (River Thurne)	Martham Sth Broad	Martham Nth Broad	Somerton Nth Pump	Somerton Sth Pump	Heigham Sound	Hickling Broad	Catfield Pump	Stubb Mill Pump	Horsey Mere	Horsey Mill Pump	Brograve Pump
	Transparency	0.46	0.31	0.70			0.31	0.28					
	Temperature	1.81	1.59	1.74	1.74	1.91	1.76	1.80	2.25	1.92	1.85	1.68	1.96
	Suspended solids			1.07	1.04	1.04		0.97	1.06	1.06	1.03	1.04	1.03
	Dissolved oxygen (% sat.)	0.60	0.97	0.86	0.34	0.41	0.88	1.07	0.51	0.50	0.88	0.35	0.63
Physical	Dissolved oxygen (conc.)	0.46	0.99	0.86	0.51		1.19	1.15	0.59	0.44	1.27	0.52	0.63
	Acidity	1.38	1.50	1.41	1.19	1.31	1.47	1.43	1.33	1.29	1.27	1.26	1.22
	Alkalinity	1.34	1.28	1.38	1.48	1.41	1.31	1.00	1.22	1.43	1.40	1.38	1.47
	Electrical conductivity	0.45	0.76	0.96	0.29	0.95	0.29	0.18	1.30	0.33	-0.37	-0.08	-0.12
Chemical	Chloride	0.59	0.92	0.90	0.35	1.32		0.31	1.38	0.44	-0.15	0.03	-0.01
	Total phosphorus	0.94	1.13	1.11	0.88	1.00	0.94	0.86	0.90		0.98	0.62	0.98
	Filtered reactive phosphorus	1.05	1.05	1.05	1.04	1.04	1.05	1.05	1.03	1.04	1.05	1.02	1.04
Nutrients	Inorganic nitrogen	1.76	1.83	1.78	1.47	1.40		1.83	-0.12	1.67	1.39	1.51	1.19
	Chlorophyll	0.55	0.86	0.86			0.22	0.26			0.52		
	Diversity of water plants		0.84	0.89				0.19			0.33		
Biological	Abundance of water plants		0.46	0.51				0.05			0.20		

A - HIGH   B - GOOD   C - MODERATE   D - POOR   E - BAD   NO DATA

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This would be combined with an assessment of costs and benefits of consuming or preserving the natural capital assets to determine the most desired outcome.[17] Alternatively, there could be a requirement to establish the sensitivity of outcomes to the choice of discount rates. If they are not sensitive, discounting practices are less of an issue. However, such approaches would be relevant only where current consumption will not breach future environmental limits.

The ultimate value of natural capital to humans is infinite, because if it is reduced beyond certain levels the human species will cease to exist. Sustaining future human welfare by maintaining levels of ecosystem services within acceptable environmental limits will require substantial enhancement of natural capital (Chapter 5). To clarify the extent to which future values placed on natural resource system benefits could be reflected in discount rates, the setting of environmental limits should reflect the likely future importance of a given benefit. For example, the need for benefits such as flood regulation is likely to increase with future climate change. This may require the application of zero or negative discount rates to cost benefit analysis of projects or policies.[173]

Marginal Costs and Environmental Limits

Economic valuation is applied not to an entire ecosystem but to an incremental or 'marginal' change within a specified policy context, as changes in value maybe specific to given location, such as a national park. What constitutes 'marginal' refers to a small part of a natural resource system, such as a land unit area, or to small changes to the overall status of the natural resource system, such as a reduction in soil organic matter.

Changes arising from larger or more cumulative impacts are more difficult to value, although there are already examples of valuations being applied at the national level in the UK. For example, techniques applied in the Impact Assessment for the 2009 Marine and Coastal Access Act estimated that the potential benefits of a UK network of marine conservation zones (POSTnote 310) would outweigh the costs of implementation between 7 to 40 times, yielding benefits of between £7 and £19 billion over a 20 year period (Section 4.12).[174]

Marginal monetary valuation of ecosystem service benefit provision assumes that it can be substituted (by manufactured capital such as technology) and does not take account of environmental limits. Valuation does not recognise that, at some point, cumulative effects on the environment mean it will become unacceptable to trade-off environmental losses against economic gains, due to the risks incurred. At this point, the uncertainty involved makes it difficult to determine how the value of a resource might change, [175] and values involved would tend towards infinity.

Safe Minimum Standards

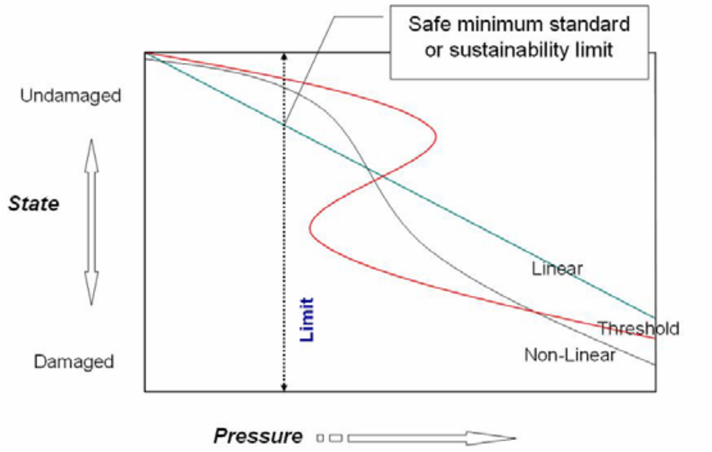
Additional safeguards are required to ensure natural resource systems do not fall below the level of ecosystem structure and process needed to maintain their functional integrity in the longer term. Safe Minimum Standards (SMS) are precautionary environmental limits relating to the boundary conditions beyond which the risk of changes in a natural resource system are deemed to be unacceptable, because the system is damaged or its integrity is threatened (Figure 15). The concept of SMS reverses the normal presumption that development is justified unless the costs to environment are very high to the presumption that conservation of natural capital is justified, unless the social and economic costs of not developing are very high.[176]

Within the decision-making process, consideration would need to be given to whether SMS would be breached under different policy options by constructing alternative future scenarios. This would include SMS for all aspects of natural capital and ecosystem services. Determining acceptable loss of natural capital requires the quantification of ecosystem services at the relevant scales, as well as societal or political judgements about the acceptable level of risk of losing those services.

Given the scientific complexities and uncertainties involved, defining SMS at a national or local level is unlikely to be a straightforward process. There is insufficient empirical evidence on natural resource system integrity and the amount of ecosystem structure and processes required to maintain the flow of services across landscapes and seascapes. Even when data about the range of consequences if particular ecosystem thresholds are crossed are available for more ecosystems, any given limit or SMS will still need to emerge from a social negotiation process, reflecting different views of risks and the possible costs involved in avoiding the SMS (Chapter 6).

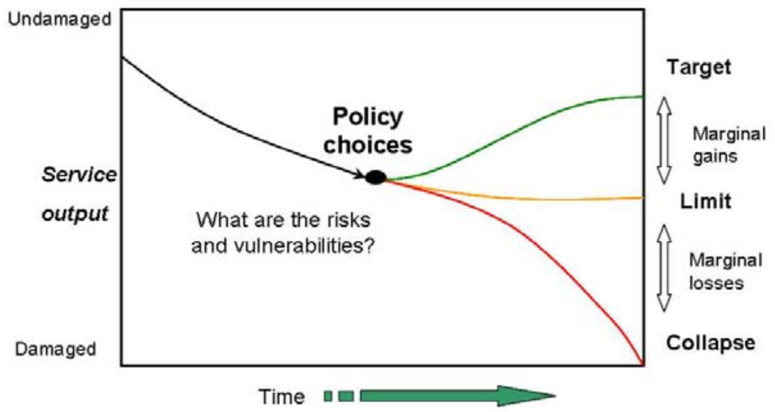
Figure 15. Safe Minimum Standards

A



A) Ecological systems or processes can show a range of responses to external pressures, which can either be linear, non-linear or threshold response where ecosystems shift into an alternative state (Section 2.6). Limits or safe minimum standards can be defined to specify the extent of acceptable change in the levels of benefits from a natural systems or the level of risk that is acceptable in relation to crossing possible ecological thresholds (Section 3.5).[177]

B



B) Policy decisions need to take into account the marginal gains in the levels of benefits delivered through steps to restore natural capital or the losses incurred through allowing limits to be breached.[177]

5 Managing Ecosystems to Ensure Resilience

Overview

- Resilient ecosystems are usually described as those that are best able to absorb disturbances, that exhibit self-organisation and that show the capacity for adaptation to environmental change. The general capacity of most ecosystems to experience shocks while retaining essentially the same function, structure, feedbacks and hence levels of ecosystem services, remains highly uncertain.
- Managing ecosystems to maximise one particular benefit, such as food provision, can result in declines in other ecosystem services. A resilient management approach to natural resource systems within landscapes would involve managing habitats to sustain the provision of multiple benefits from ecosystem services, such as food, production of biofuels and fibre crops, and protection of sources of pollinators and species that control pests.
- A fundamental aspect of maintaining the resilience of natural capital assets and the flow of public benefits from them are the regulations applied to, and the payments made to, owners and users of natural resources. The evidence base to show how management measures could increase the resilience of ecosystems to environmental change and maintain or improve benefit provision is not yet sufficient, particularly for direct payments for ecosystem services from users to suppliers.
- The implementation of adaptive management systems and effective monitoring of changes in ecosystems will be critical to determining the most effective measures for increasing system resilience, and to avoiding exceeding environmental limits when abnormal events occur, for many natural resource systems.

5.1 Natural Responses to Environmental Change

Environmental change alters ecosystem properties through effects on the physical and biochemical environment and through direct effects on the biology of organisms. These effects have consequences at the species and communities level affecting ecological processes in ecosystems. All ecosystems have experienced environmental change and disturbance over geological millennia. Long-term records of ecological history from peat, and from lake and marine sediments show that change is normal and that ecosystems have adapted to environmental change in the past.

This is reflected in the process of ecological succession, in which, following the disturbance of a habitat, for example, by a fire or flood that destroys most of the organisms present, distinct communities of organisms will colonise a site in succession. Primary succession occurs on bare or recently uncovered surfaces, such as mud or bare rock where soil has yet to form. The replacement of a plant community by another distinct plant community on a well-developed soil is called secondary succession, for example, when agricultural land is abandoned and progressively becomes reforested.

Disturbance refers to any relatively discrete event in time that disrupts ecosystems or which drives changes in the natural resource system or in the physical environment.[178] Natural disturbance events can be valuable agents in creating successional cycles on which communities of organisms depend, such as canopy gaps in forest ecosystems. Such processes rely on there being a local pool of species with sufficient dispersal abilities to be able to re-colonise a site.

However, if species become locally extinct due to lack of available habitat or environmental conditions change to the extent that species cannot physiologically adapt, impeding reproduction and growth (Section 2.5), then recovery from disturbance becomes unpredictable.[156] Most semi-natural habitats in the UK, such as grasslands or heathland, were created by particular forms of human land-use, often over millennia, and depend upon ongoing management, such as grazing or cutting, to prevent succession and the loss of benefits for which that habitat is valued.

Resilience and Resistance

In the ecological literature, resilient ecosystems are generally seen as those that are best able to absorb disturbances, that exhibit self-organisation and that show the capacity for adaptation to environmental change. Theoretically, the degree of resilience of an ecosystem may be determined by:

- The ability of the system to return to normal after disturbance or stress such as, the recovery time for the vegetation community following a fire or flood. The more resilient an ecosystem is, the shorter the time it takes to do this.
- The magnitude of disturbance or energy that can be absorbed before the system alters its structure by changing the variables and processes that control behaviour of the system. Examples are the amount of fish biomass that can be harvested before a marine ecosystem changes to a less favourable state, such as dominance by species lower down the food chain, or the frequency of crop failures in agricultural systems.

This latter criterion is sometimes categorised as "ecological resistance" rather than resilience,[179] or as a subcategory of resilience. Resistance directly relates to environmental limits, as it is the amount of disturbance that can be tolerated before biophysical thresholds are breached (Section 2.6). Functional diversity, the component of biodiversity that concerns the range of things that organisms do in communities and ecosystems, is important in maintaining processes in ecosystems (Annex B),[180] and a diversity of different species performing similar roles allows for faster recovery from disturbance (resilience).

Research suggests that genetic diversity within species that perform important roles within ecosystems is also important for resistance to abnormal events. For example, eel grass (*Zostera marina*) forms an important habitat for marine species in shallow estuaries in temperate areas. Experimental manipulations of eel grass plots grazed by geese have shown that the amount of genetic diversity within the stands of the grass is critical to determining whether they were able to resist grazing disturbance.[181]

While resilience and resistance characteristics are in principle measurable through direct observation in ecosystems, other relevant attributes such as the capacity of ecosystems to transform and adapt to environmental change remain poorly understood.[182]

Resilience and Biodiversity

The sparse data available suggest that those ecosystems that retain high levels of functional diversity to act as 'insurance' are most able to withstand multiple or repeated disturbances (such as sudden changes in nutrient levels or an invading species, Annex B) and recover faster (both more resilient and resistant). There is growing scientific concern that biodiversity loss may lower resilience and resistance to disturbance.[183]

The ability to predict how an ecosystem will respond to disturbance and environmental change remains a major challenge for ecological science. Not enough is known about the different levels of resilience and resistance of ecosystems in given different states, and how change can be successfully managed to maintain levels of benefits important to human well-being. There is evidence that species richness and more diverse patterns of species interactions can enhance ecosystem state stability and maintain levels of ecosystem services.[184]

It is unlikely that particular thresholds for biodiversity loss, or other aspects of changes in natural capital, can be determined with the certainty necessary to predict unacceptable ecosystem changes and to set environmental limits. It cannot be assumed, for instance, that the historic ability to absorb repeated shocks by ecosystems in the UK will be maintained or that present levels of ecosystem services will continue, particularly with increasing pressure on biodiversity from climate change (POSTnote 341). Most commentators agree these uncertainties highlight the need for a precautionary approach to managing natural resources and related ecosystem services to avoid breaching thresholds.

Regulating Services and Resilience

Interaction between regulating services, such as pollination services and carbon sequestration, may be critical in ensuring the maintenance of other ecosystem services, but this has yet to be systematically assessed.[185] Declines in regulating services could result in greater fluctuations in the provision of other ecosystem services, such as food protection, as well as decreasing the resilience of ecosystems to environmental change.

Abrupt shifts in ecosystem states are often triggered by external shocks, such as invasive species or a drought, but the resilience to such shocks can be enhanced by restoring regulating ecosystem services. For example, management measures that support regulating services, such as farming methods that reduce soil erosion, increase the resilience of agricultural ecosystems. By contrast, measures to enhance provisioning services, such use of fertilisers and pesticides, may lead to declines in other ecosystem services and the resistance of agricultural ecosystems to environmental change.

Regulating services are usually linked to “slow changing variables”, which are critical to whether ecosystems shift to another state. These variables change on timescales slower than the dynamics of other ecological processes they support.[186] For instance, in the case of a shift from coral reefs to algae-dominated reefs, the regulating service of algae-eating fish declines dramatically over a period of time before the shift to an algae-dominated reef actually happens.[187]

In agricultural ecosystems, reduced soil organic matter, a key “slow variable”, can lead to decreased water-holding capacity. Less water in soil reduces its capacity to cope with a high frequency of dry spells.[188] For example, in East Anglia, due to tillage and drainage over decades reducing organic matter in the soil, the resilience to a high frequency of dry spells has been reduced. However, further research is needed to clarify how abrupt shifts in ecosystem states are related to interactions between regulating and other ecosystem services.

5.2 Social Ecological Systems

However, environmental policies cannot be decided simply on the basis of scientific evidence. Environmental limits are a reflection of how society places value on the ongoing provision of particular benefits arising from an ecological process, and the loss of such benefits will reduce social well-being. There is now a significant body of academic literature that suggests that rather than considering resilience to be solely a property of ecosystem biophysical thresholds, it should be defined within the context of the interactions of physical, ecological and social processes.[182]

These are often referred to as “social-ecological systems” (SES), consisting of physical components (soil, water and rocks), organisms (plants, microbes and animals, including humans), and the products of human activities (food, buildings and pollutants). As with ecological processes, such systems can be defined at different scales, from the household and community level through to the national and global level.[189]

Resilience within the context of such “socio-ecological systems” is often defined as “...the capacity of social-ecological systems to absorb recurrent disturbances…so as to retain essential structures, processes and feedbacks”.[190] Although this is a much less precise definition than the ecological concept, it allows for a more holistic policy perspective that identifies the range of ecosystem attributes that might be promoted through ecosystem based management to sustain benefits from ecosystem services.[182]

Management of Natural Capital for Resilience

Humans often manage ecosystems to maintain a particular successional stage, such as grassland or forest, to deliver particular ecosystem service benefits. Many important landscapes in the UK need to be managed in this way, such as chalk grassland or heathland. However, successional stages will differ in their response to environmental change. Humans are increasingly playing the role of ecosystem engineers to maintain ecosystem states, for example by culling species, such as deer to avoid overgrazing pressure, following the elimination of their natural predators.

It cannot be assumed that systems previously exploited to support human well-being will continue to function within the boundaries that they have previously done. If natural systems cannot be managed to maintain their current status, the functional properties of ecosystems important for human well-being will need to be defined. This is in order to maintain the underlying capacity to deliver desired ecosystem services in the face of a fluctuating environment and human use.

Policymakers are increasingly interested in the limits of ecosystem functioning so that they can be sustained in spite of drivers of environmental change. The identification of these ecological thresholds depends on knowing how ecosystems react to, and buffer, these external pressures. This ability is usually referred to as ‘resilience’, although it includes resistance, resilience and adaptation.[182] Resilience approaches focus on the variables that underlie the capacity of SES to maintain provision of ecosystem services.[191]

It has been suggested that erosion of any of four distinct dynamic properties of an SES, based on the extent to which disturbance is transient or chronic and whether it is external or internal to the system, will determine the risk of an ecosystem crossing an ecological threshold (Figure 16). Within this framework:[192]

- A resilient ecosystem would be able to maintain functions through transient and external shocks and quickly resume ability to yield an ecosystem service when it returns to stability.

- Ecosystem stability refers to the ability to tolerate transient and internal shocks, often through feedback mechanisms, so that the system tends to a steady or dynamic equilibrium state.
- An ecosystem is robust when it is capable of resisting changes caused by chronic external pressures, such as climate change. Robust ecosystems can adapt to changes, such as the loss of a species that is critical to ecosystem functioning through other species compensating for the loss of function.
- A durable ecosystem copes with internal chronic stress and continues to provide an ecosystem service without any degradation of the components that make up the ecosystem. For example, evolution is an example of an internal pressure that acts on the biological components of an ecosystem.

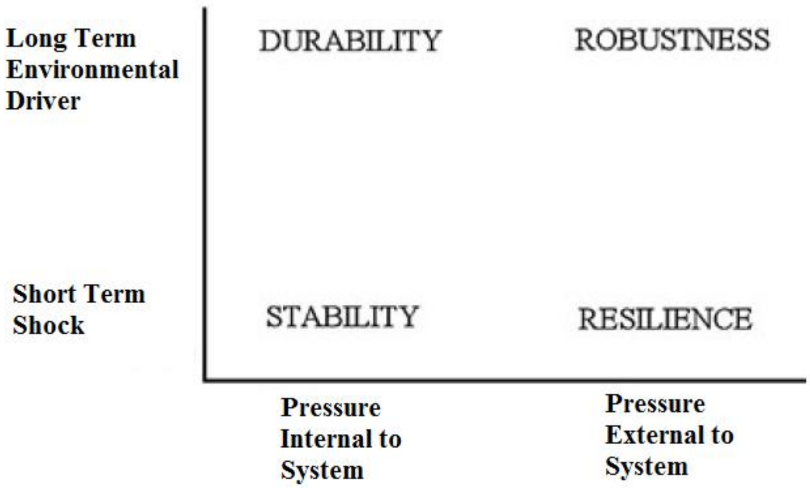
Natural resource systems are often managed to minimise the likelihood of minor disruptions such as storms, fire and flooding, to increase stability of the system. However, natural disturbance events can increase the resilience of human and environment interactions. For example, many traditional land management systems have evolved over millennia to be highly resilient resource management systems with adaptive strategies for maintaining the provision of ecosystem service benefits in changing and uncertain environmental conditions, such as use of drought or disease resistance species in agriculture.[193]

By contrast, modern intensive agricultural practices optimise inputs and outputs, to increase productivity per hectare through technology, breeding and increased energy inputs. If the four system properties set out in Figure 16 are applied, intensive farming systems appear to increase stability but at the expense of durability and resilience, and possibly robustness, compared with traditional land management systems.[194]

Although humans usually manage landscapes to maximise delivery of one particular benefit, such as food provision, interactions between ecological processes link the production of one ecosystem service benefit to another.[195] The interaction can be in one direction, with increases in the provision of a benefit affecting the provision of another, or in both directions, with the changes occurring in a cyclical nature. A one-way directional effect is demonstrated by the retention of forest patches near coffee plantations, which increases levels of pollination services, increasing coffee yield, but levels of coffee production do not directly impact on pollination services, although if they involve greater application of fertiliser and pesticides there can be indirect negative impacts (Box 2).[196]

Key challenges are to identify the factors that can enhance interactions that increase system stability and durability and how management interventions may result in greater resilience to external pressures.[182] Ecosystem based management measures could help to sustain ecosystems impacted by drivers of environmental change or to protect the capacity of ecosystems to recover following disturbance (resistance and resilience). It is probable that management that attempts to maximise the provision of a single ecosystem service increases the risk of exceeding environmental limits.

Figure 16 A Framework Representing the Four System Properties of Resilience[197]



The properties are defined according to the temporal nature of the pressure and its origin. For example, in the example of a long term external driver such as climate change, robust ecosystems will adapt without loss of ecosystem services without impact human well-being

For instance, in the cloud forests in Chile, at a particular level of tree cover the moisture found in fog is retained, improving tree growth and enhancing carbon sequestration. Cutting the forest for timber provision not only directly reduces rate of carbon sequestration, but by reducing moisture retention and growth rate of the remaining trees, further reduces carbon sequestration and future timber provision (negative feedback). The moisture intercepted by the vegetation allows the vegetation to persist, but if the tree cover, and hence total leaf area, is reduced by deforestation below a certain level, insufficient moisture is captured to sustain vegetation cover, resulting in a shift to more arid ecosystem types such as shrubland.[198]

Use of Technology and Resilience

The use of technology to eliminate or control the effects of disturbance on natural resource systems to maintain the flow of benefits can decrease resilience. For example, if an area is regularly flooded this will either discourage house building on a significant scale or encourage the building of houses resistant to the effects of flooding. However, if flood defences to a one in a hundred year standard (a level of flooding that has a one in a hundred chance of occurring in any given year), are implemented, to reduce the frequency of flooding events, development will be encouraged, resulting in greater social and economic vulnerability to a one in a thousand year flood.

Although the ecological processes in ecosystems determine levels of ecosystem service benefits, prevailing technology influences how humans can exploit ecosystems to enhance delivery of particular benefits. The interaction between aspects of the ecosystem and available technology will determine resource use patterns and will also determine the state of the ecosystem and the level of other ecosystem service benefits supported by that ecosystem. Technological innovation has increased the benefits derived from ecosystems, and these gains in efficiency have expanded the size of economies and further increased use of ecosystems, raising the risk of abrupt changes in ecological processes. However, reducing the human use of ecosystems could result in a decline in human well-being.

Any change in the provision of ecosystem service benefits may also be a limit in terms of the ability of economic and technical systems to adapt to the change. For example, the economic cost of measures to adapt to the change may exceed resources available[199] or the technology needed to adapt does not exist. As with ecological thresholds, these limits are absolute and objective.[200]

Social Systems and Environmental Limits

There are also social limits to adapting to changes in relation to the way societies are organised, the values that they hold and the relationships that exist between individuals, institutions and the state. These limits are mutable, subjective and socially-constructed. This creates a problem in defining environmental limits which reflect both objective and absolute scientific and economic criteria as well as more mutable social criteria.

When an ecological threshold is crossed, natural resource systems are unlikely to regain their original state without significant intervention. Although exceeding ecological thresholds may represent an absolute limit to the existing level of ecosystem service benefit provision, social systems may be able to adapt to this change. However, the level of adaptation may not be reconcilable with social values. An example is abandoning large areas in response to sea level rise (POSTnote 363).

If subjective social values are not in accord with objective scientific and economic criteria, the ability of socio-ecological systems to adapt to environmental change will be reduced. The diversity of values may lead to a paralysis of measures to maintain natural resource systems, such as the failure to introduce or change regulatory incentives. It can also lead to contradictory outcomes, such as management measures which simultaneously yet differentially enhance and reduce resilience to environmental changes.[200]

Landscapes are dynamic social constructions that are the result of complex interactions of cultural, political and ecological processes, and they can be culturally important.[201] Local institutions, such as local planning authorities, are critical as to creating resilient landscapes that can adapt to change (Annex D). At larger geographic scales, policy decisions such as the 2013 revision of CAP will also affect human use of natural resource systems. Without responsive and adaptive institutions, environmental change can lead to the exceedance of environmental limits and to social and economic crises.[202]

Resilient Natural Resource Systems

Recent studies have identified possible system variables that affect SES, focussing on the interplay between resource and governance systems, resource units and users to identify policies, rules, monitoring and enforcement strategies.[203] Diverse human actors respond to changes in ecosystem service benefit provision, environmental factors or social factors and influence the response of regulatory institutions to these changes (Figure 17).

Whereas institutions and markets can shape the way that individuals interact and use natural resource systems, the status of natural resource systems determine the quantity and quality of ecosystem service benefits that are potentially available to society. Changes in the dynamics of SES are regulated by the interaction of slow variables (such as changes in regulating ecosystem services, for example, loss of soil fertility) and fast variables (such as disturbance by flooding).

Complex Adaptive Systems

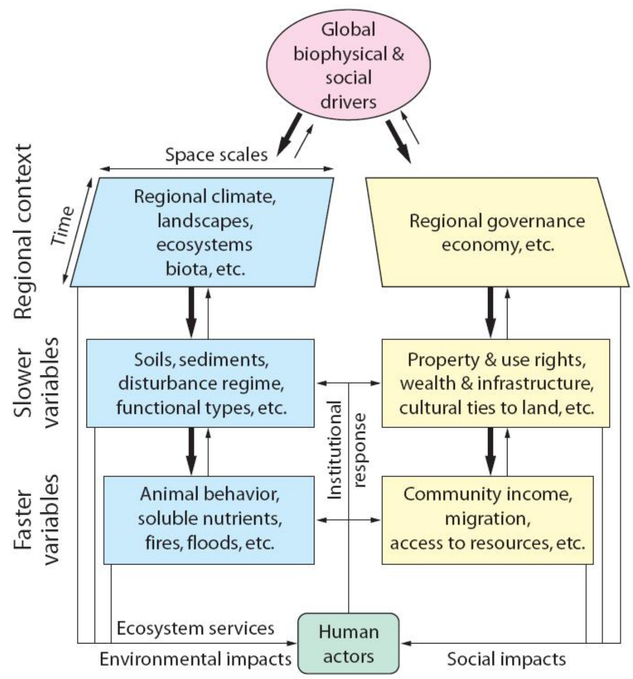
Complexity theory was developed from key ideas in economics, physics, biology and the social sciences. SES are a form of Complex Adaptive System (CAS) in which a dynamic network of many agents acting independently are in parallel, constantly responding to their environment and to what the other agents are doing.[204] CAS tend to be highly dispersed and decentralised and can exhibit emergent behaviour and self-organisation, phenomena that cannot be predicted directly from the properties of their component parts.[205] To improve understanding of complex system behaviour, modelling is required to determine the range of emergent system behaviours of any given social ecological system.

For example, farm viability is an emergent property of a complex network of social, environmental and economic interactions. Complex systems are large aggregations of many smaller interacting parts. Two properties that set a CAS apart from one that is merely complicated are those of emergence and self-organisation (where a structure or pattern appears in a system without an external element imposing it). Emergence is the appearance of behaviour that could not be anticipated from knowledge of the parts of the system alone. Other examples of complex emergent behaviour in SES include the collapse of fish stocks and the over-nutrification of waterways outlined in previous chapters.

In systems theory, the greater the density of links or interactions or the greater the intensity of interactions within the system, the greater is the risk of instability. However, in the case of ecological systems, components of systems have co-evolved over long periods and it appears that diversity increases resilience and buffers against external shocks (Section 2.7). By contrast, as SES increase in complexity and connectedness, they may increase the likelihood of collapse as all ecosystems become human dominated and simplified.[206]

Some research studies have also suggested that as social systems become more complex and interconnected, they are less able to adapt to ecological and environmental change. As societies become more complex, with divergent values and intricate forms of governance, investment in

Figure 17 One Possible Framework for the Analysis of Social-Ecological Systems.[9]



problem solving activities and technologies may deliver less resilience.[207] The MA considered the issues around social-ecological systems to some extent, but further understanding of the interactions of social and ecological constituents at different spatial and temporal scales is required.

Vulnerability Approaches

Environmental policies have previously been based on vulnerability approaches. These focus on the most endangered biodiversity, or on social groups most at risk, and seek to implement measures aimed at their protection. Management to reduce vulnerabilities is likely to include priorities other than resilience. It could actually reduce resilience, as protecting the most vulnerable, such as communities heavily affected by coastal erosion, may be at odds with creating resilient socio-ecological systems.[208]

Without appropriate monitoring to inform policymakers about feedbacks from decisions and potential new risks arising from changing social, economic and ecological conditions, vulnerability approaches can fail to predict emerging vulnerabilities in sufficient time to manage resulting risks. However, there is also a potential for measures to increase resilience to be perceived as disbenefits, such as the setting aside of areas near communities for flood storage (Box 15). Resilient socio-economic systems require both the need to reduce the demand for benefits that drive pressures on natural resource systems and acceptance of management measures.

Whether resilience, or vulnerability, or risk-based approaches are adopted as a policy response to environmental change, there will be a range of hidden assumptions and trade-offs. While appearing to be the most efficient and appropriate response to environmental change, the resilience approach of prioritising long-term public good over immediate private gains or the welfare needs of societal sectors is politically challenging (Section 6.2). A key policy concern will be to ensure social justice issues are addressed through open debate, such as participatory approaches (Section 4.10), and the explicit clarification of priorities associated with alternative policy choices.[208]

Building Resilience

Resilient natural resource systems require the use of the resource to be balanced against the need to maintain the ecological processes that underpin benefit provision. A resilient approach to managing natural resource systems within landscapes would involve managing habitats to sustain the provision of multiple benefits from ecosystem services under a range of pressures or stresses.

Resilient systems should be able to cope with abnormal events placing transient stress on the system (Figure 16). For example, water scarcity is likely to become a normal state in South East England and water supply systems will need to work within these limits as matter of course, but they will also need to be able to cope with abnormal droughts (and floods) over and above this. At present, increased demand is making the water system less durable and less resilient.

Building-in the resilience to abnormal events beyond the broad band of normal events does not mean designing systems resistant to constant exceedance, as this would be too resource intensive. However, there should be sufficient storage or reserves of natural resources to facilitate rapid recovery from events. Ecosystems should be maintained in a state that allows for recovery from disturbance (ecological resistance), and for the provision of benefits to be maintained (within environmental limits).

If abnormal events cause an ecosystem to cross a biophysical threshold (ecological resilience) then environmental limits will have already been exceeded, even if benefits had been maintained up to this point. Maintenance of provision of benefits at acceptable levels in the future will have been compromised by failure to maintain sufficient capacity for recovery. This situation is observed in the collapse of the Canadian Grand Banks and other fisheries across the world, where increased technology maintained yield up to a point of total collapse.

Most management systems operate within a design capacity. For example, farm irrigation systems are designed for the fifth driest year in twenty as an acceptable level of risk. Of course, this year may become drier, with events previously regarded as extreme becoming more common. This may require a rethink of the whole system to increase durability and robustness. Over-use of natural resource systems in the long term increases the risk of system collapses, regardless of the steps are taken to cope with abnormal events.

Resilience approaches would focus not on specific risks, but on developing the capacity of the whole SES to cope with change and uncertainty through maintaining flexibility, trading off economic efficiency against long-term functioning. However, at present there is insufficient understanding of SES to be able to develop policies that enhance their resilience to transient environmental shocks and their robustness to withstand long term environmental pressures, such as climate change.

Resilience Accounting

If there were no uncertainty about the dynamics of a natural resource system, it could always be managed to stay within the bounds of resilience. However, the reality is that there will always be uncertainty and a risk that the system will reach a biophysical threshold and move to the non-desired state. This risk will be lower if the resilience of the system is increased. Furthermore, resilience could be regarded as a capital stock (Section 3.1), as it provides insurance against reaching a non-desired state. As such, it has an economic value that is roughly the change in net present value of the expected future ecosystem services resulting from a marginal change in resilience today.[209]

For example, resilience values have been calculated for the Goulburn-Broken Catchment in Southeast Australia, an area with extensive production of vegetables. Irrigation has increased the salinity of the ground water so that, if the water table rises to two metres under the surface due to multiple consecutive wet years, the saline water would be sucked up to the surface and the whole production of agricultural crops would collapse. The largely historical clearing of native vegetation particularly in the catchment foothills, combined with the extensive development of irrigation infrastructure, and reliance on pumping systems to control water table levels has over time reduced the number of options available to reverse this trend.

Thus, the resilience in this system is the distance from the current level of the water table to two meters below the surface. Based on historical data (which may no longer be relevant because of climate change), the researchers estimated a probability distribution for the level of the water table and then estimated the increase in expected net income from the agriculture if the water table were lower, to represent the effective capital stock of resilience.[210]

5.3 Enhancing the Resilience of Ecosystem Service Provision

There is evidence about how individual habitats interact to give rise to the processes that support ecosystem service provision. Research studies are taking place at the farm level to determine the optimum state and mix of habitats to support ecosystem services. These could be scaled up to larger areas. Less is understood about how much good quality habitat is required and where it should be located within a specific landscape or marine area to ensure sufficient resilience to external transient shocks such as droughts or invasive species.[211] However, there is evidence that landscapes dominated by intensive arable monocultures are more vulnerable to shocks than ones with greater proportions of semi-natural habitat.

Management of most natural resource systems also tends to change only when a problem or significant risk, such as increased flooding or loss of a valued species (Box 17), is apparent and resilience is impaired. Key management issues for restoring degraded natural resource systems include:[212]

- The ecological interactions and processes that most affect the changes in provision of benefits, considering the state of local ecosystems.
- The current structure and trends in the natural resource system (established through an ecosystem assessment, Section 4.7).
- The tools or uses, such as agricultural or fishing practices that have altered the system to deliver particular benefits.
- Control of resource use (property rights, Section 5.9) and regulations at the relevant scale.
- The intensity of use to produce a particular benefit, such as agricultural productivity, which will not have significant negative impacts on maintaining the provision of other key benefits.
- The optimal level of provision of different benefits (informed by valuation and public engagement, section 4.8) and the arrangement and state of habitats or ecosystem structures required to deliver and maintain this.
- The tools that would alter the system to deliver the optimal level of provision of different benefits (such as agri-environment schemes or coastal retreat).

For example, between 1990 and 2007 there has been a reduction in the total area with wild nectar plants available for supporting pollinators in Great Britain. This is mainly because they have been crowded out by the growth of more competitive plant species, due to changes in land management and increased levels of nitrogen deposition from air pollution.[135] The UK Biological Records Centre holds data on pollinators at the 10km^2 level which could be used to identify areas where there is the lowest diversity of pollinator species and consequently a greater risk of catastrophic regime change and loss of pollinator service provision.

Box 18 Chimney Meadows National Nature Reserve[213]

Wetlands cover less than 3% of the terrestrial surface but are estimated to contribute 40% of ecosystem service benefits, including carbon storage, water quality and flood control. Preservation of wetlands in the UK is critical to maintain ecological infrastructure and ensuring resilience in levels of ecosystem service benefits.

A typical example of lowland wetland is the Chimney Meadows National Nature Reserve in the upper Thames catchment. The type of floodplain grassland habitat found on the reserve, meadow foxtail and great burnet grassland (referred to as Magnesium Grassland 4 using the National Vegetation Classification of habitats) is maintained through a management regime that includes regular winter flooding. This promotes an exchange of nutrients between the grassland habitat and the river.[214] Throughout the year, the balance between the water regime, nutrient regime (such as the input of nitrogen and phosphorus from flooding) and the management of the vegetation through cutting is critical to maintaining the community of plants and animals found in the habitat.

The importance of this balance was shown by the impacts of summer flooding in both 2007 and 2008, which greatly increased deposition of nutrients.[215] This led to a decrease in the diversity of plant, invertebrate and bird species present in the grassland for which the reserve was designated. Changes in management, such as hay cutting followed by grazing, may reduce levels of nutrients over a number of years.[216] Flooding also affected the soil macro-fauna through soil water-logging with worm density falling by 63% across the reserve. Increased flood risk is also being addressed through the inclusion of integrated wetland features in the site, such as ponds, scrapes and dykes .

The reserve has been identified in the Thames Catchment Flood Management Plan as a flood storage area. However, if flood storage becomes the main driver for management, this will conflict with the requirement to maintain the hydrological regime, which is key to conserving the biodiversity and associated ecological processes within the habitat. Further reductions in community complexity will increasingly erode the resilience of the habitat to additional perturbations, such as unseasonal floods, until a threshold is reached and the existing community of plants and animals is replaced with a different ecosystem state. To maintain the ecosystem service benefits associated with the reserve, it could be recreated on adjoining habitat that can be more easily protected from unseasonal flooding. However, to avoid widespread loss of species associated with riparian (river or stream bank) and aquatic habitats, a more strategic approach is required to restore floodplain function and reduce the impacts of unseasonal flooding to enhance or maintain the resilience of such habitats.

Once identified, there are three possible options for increasing the resilience of pollination services in high risk areas:

- Importing managed pollinators (honey bees, bumblebees or mason bees) into the area to increase pollination rates. Of the available options, this is short term and unsustainable, as managed pollinators can pollinate only a limited range of plant species, and some at less efficient levels, than the full range of wild pollinators.
- Creating a fixed percentage of wild flower meadows and other habitat structures needed to support populations of wild pollinators per km^2 of landscape. This relatively untargeted approach may not result in the optimal habitat arrangement within a landscape for pollinators but is the response that can be implemented most immediately.
- Full scale ecological engineering of landscapes to have an optimal mix of habitats to deliver multiple ecosystem service benefits, including pollination services.

Only the last of these options will deliver sufficient resilience to assure that environmental limits are not exceeded, but there needs to be better understanding of how interactions between habitats occur at different scales before it can be determined how heterogeneous a given landscape should be. Past patterns of habitats may provide information on how previous uses of habitats supported ecosystem service benefit provision in landscapes. However, the appropriate point at which to delineate a historical baseline is debatable for many ecosystem service benefits. These have been gradually declining over a long period in response to a range of pressures and many historical changes in landscapes.

Mapping landscapes to determining existing natural and semi-natural elements would be the first step in determining the features critical to maintain ecological processes. However, it may well be necessary to restore or recreate habitats where they are lost due to land use change, or degraded due to over-use of a particular resource, such as abstraction of water or intensive agriculture. However, changes in the way land, freshwater or marine ecosystems are managed may often be contentious, with both the direct users of the resources and those affected more indirectly.

Ecological Restoration

Ecological restoration involves assisting the recovery of a habitat that has been degraded, damaged or destroyed, usually as a result of human use of those habitats. The main aim of restoration is to re-establish the characteristics of a habitat, such as biodiversity and ecological processes, that were prevalent before degradation.

This involves removal of the drivers of degradation and re-establishment of key ecosystem components, such as the hydrological regime in wetlands. In marine ecosystems, it is usually possible only to remove the drivers of degradation, such as excluding damaging fishing activities from an area, but studies are ongoing on techniques such as coral reef restoration and mangrove re-introduction.

Any management approach to enhancing the resilience of natural resource systems is likely to involve restoration of degraded habitats (such as hydrological restoration of drained wetlands), the halting of inappropriate uses of ecosystems (such as forestry plantations on peat bogs), and the expansion of existing semi-natural habitats and 'buffer strips' between damaging uses and semi-natural habitat (ecological networks). For example, the Great Fen Project will create 3700ha of wetland between Huntingdon and Peterborough, joining up two National Nature Reserves. The project will restore a mosaic of habitats including fens, wet grasslands, reedbed and woodland. The restoration of the wetlands will prevent an estimated loss of 325,000t of carbon dioxide into the atmosphere annually, and will increase levels of biodiversity and provide tourism and recreation opportunities.

Habitat creation and restoration is already carried out or funded by a wide range of organisations. Thus includes government departments and agencies, voluntary conservation organisations, the Heritage Lottery Fund, grant-giving charities, landfill tax bodies, businesses and private landowners. Although management measures to restore ecosystems services are poorly developed for some services, and more evidence is needed on which are the most effective, there is the potential for costing out restoration at the level of habitat units.

Delivering ecosystem restoration measures on a large scale can be achieved through:

- direct government payments to landowners (agri-environment schemes);
- creating a market for ecosystem service provision to finance restoration (through a system of habitat banking or payments for ecosystem services);
- extending tax incentives to encourage the creation, improvement and long-term maintenance of wildlife habitats out of private resources; and,
- regulating how land, sea or inland water bodies are managed to maintain the flow of ecosystem services (Section 5.9).

In practice, it is likely that restoration of terrestrial, inland water and marine ecosystems to re-establish ecological processes and to maintain the provision of key ecosystem services, will be delivered through a mixture of all these measures. Although the level of service from restored ecosystems remains below that of natural ecosystems, studies indicate that restoration can give economic rates of return. Overall, however, avoiding degradation is more effective than restoration for maintaining resilience.

Evidence Base for the Effectiveness of Restoration Measures

A meta-analysis, a statistical method of combining evidence from different studies, of 89 ecological restoration assessments in a wide range of ecosystems across the globe indicated that ecological restoration increased provision of biodiversity by 44 % and ecosystem services by 25%. The research showed that while undamaged habitats are best for ecosystem service provision, restored habitats can provide some services, whereas damaged habitats provide the lowest levels of ecosystem service benefits.

Restored ecosystems are unlikely to be a reproduction of original ecosystems where many species have been lost, as many of the original components at the micro-organism level, although often critical to ecological processes such as nutrient cycling and soil structure, are not known.[217] It is likely that combinations of species that never previously existed together will occur and there will be inherent uncertainty as to whether ecological processes can be restored once a habitat has been lost. For example, total duplication of natural wetlands is impossible due to the complexity of the relationships between hydrology, soil, vegetation, animal life and nutrients which may have developed over thousands of years.[218]

A key habitat for restoration to support ecosystem services in the UK is peatlands, which are the UK's largest terrestrial carbon store. Restoration can safeguard these stores, reduce emissions from them and potentially enhance long term sequestration. In addition, upland peatlands are source habitats for about 70% of UK drinking water, and restoration could contribute to freshwater quality and aid flood mitigation. Current projects to restore peatlands in the UK include the Exmoor and Dartmoor "Mires.on the Moors" project, "Flow Country", the "Lake Vyrwy LIFE project", "Moors for the Future", "Peatscapes", "SCaMP" and the "Yorkshire Peat Project". However, the evidence base to support the contention that restoration provides multiple benefits remains inconclusive and further demonstration projects and long-term monitoring experiments are required to supply corroborating data.

Ecological restoration restores natural capital and improves ecosystem service provision, but the optimal mix of habitats at the landscape level to maintain ecosystem service benefits is not well understood.[219] Attempting to restore habitats against a background of environmental change may also be challenging.[220] A recent review for Defra concluded that restoration needs to take place throughout

England, with areas where restoration activity should be concentrated referred to as Ecological Restoration Zones (ERZs). These are areas characterised by:

- A shared vision for an enhanced, resilient natural environment existing among local communities, landowners, local authorities, NGOs and government agencies.
- Significant enhancements of the ecological network over large areas, planned by enlarging and enhancing existing wildlife sites, improving the ecological connectivity between sites and/or creating new wildlife sites.

ERZs are being proposed by consortia of local stakeholders supported by national agencies with funds made available on a competitive basis to implement 12 ERZs.

The economic valuation of ecosystem services could act as basis for restoration-based credits in an environmental market to fund ecosystem restoration (a form of "biodiversity offsetting"). However, devising methods to assign economic value or mitigation credits to an ecosystem service benefit, such as water quality, does not necessarily mean a service will be restored.[221] Not enough is understood about the ecological interactions between habitat patches at the landscape level that support ecosystem service benefits. In addition, habitats, once altered, cannot be restored to provide the full suite of ecosystem services they would have once supported.[222] Some habitats, such as ancient woodland and limestone pavement, are also effectively impossible to recreate within human timescales.

Ecological Networks

The majority of existing good quality habitat in UK landscapes has some form of statutory protection, and could form the basis of an expansion of good quality habitat through restoration. However, the distribution of existing habitat may not be optimal for provision of ecosystem services or possible future environmental changes. In addition, small isolated patches of semi-natural habitat within intensively used landscapes may be unable to support many species populations in the long term, because they do not contain sufficient resources or habitat diversity to sustain them.

Species of plants and animals differ enormously in the size of their home ranges, seed dispersal distances, population densities, and their ability to cross hostile landscapes. Isolated fragments of habitat, and the species within them, have less resilience to abnormal damaging events, such as wild fires or more gradual environmental change as any species they contain are less able migrate between them. For example, in England, most wildlife sites are small, with 77% of SSSIs and 98% of Local Wildlife Sites less than 100ha in area.

However, restoring large expanses of continuous habitat is not a feasible option in much of the UK. Some studies suggest that reducing the overall land use intensity and either improving the quality or size of remaining semi-natural habitat patches,[223] can increase their resilience. It is almost always the case that large areas support more species than smaller areas because they are more physically variable, providing greater habitat diversity, and because they contain larger populations of individual species that are less likely to fluctuate to local extinction (for example, due to a cold winter).

As an alternative to continuous habitat, it has been suggested that "ecological networks" could be used to maintain ecological structures and processes across landscapes (POSTnote 300). These are a suite of high quality semi-natural areas which collectively contain the diversity and area of habitat that are needed to support species and which have the ecological connections between them that enable species, or at least their genes, to move between them.[57] These connections can be in the form of appropriately managed buffer zones around a high quality site, wildlife corridors or small patches of habitat remaining between them. Creating a patchwork of habitats throughout a mosaic of mixed land uses may also be an effective means of maintaining ecological processes.[224]

There is evidence that habitat patches within a fragmented landscape that are connected have higher levels of biodiversity than those that are not,[225] indicating that existing connections should be maintained. It is possible to compare the levels of fragmentation and connectivity, taking into account the size and distribution of remaining habitat patches and the quality of the landscape matrix, as shown for England in Figure 18. Relatively mobile groups of species such as butterflies, birds and large herbivores, have been shown to benefit from landscape features between habitat patches. However, the evidence base for whether the connections between habitat fragments can be effectively restored through recreated habitat remains limited for most other groups of species.

Defra recently commissioned a review of how a coherent and resilient ecological network could be created in England from existing wildlife sites with statutory protection. The review concluded the current set of sites does not comprise a coherent and resilient ecological network that is capable of coping with drivers of environmental change, such as climate change. To be coherent, the network components need to be complementary and mutually reinforcing, so that the value of the network is greater than the sum of its parts. To be resilient, the network should be capable of absorbing, resisting or recovering from disturbances and damage caused by natural perturbations and human activities, while continuing to provide ecosystem services.[57]

The annual costs of implementing a coherent and resilient ecological network were estimated to be between £600 million and £1.1 billion, as compared with a current expenditure of £400 million to meet Biodiversity Action Plan objectives.[57] Natural England has recommended a sequence of actions to improve the resilience of existing habitat, where the creation of new networks to connect habitats occurs only after other suitable actions have been taken to strengthen the conservation of existing sites and to buffer them against threats.[226] The review recommends that local authorities should ensure that ecological networks, including areas for restoration, are identified through local planning.[57]

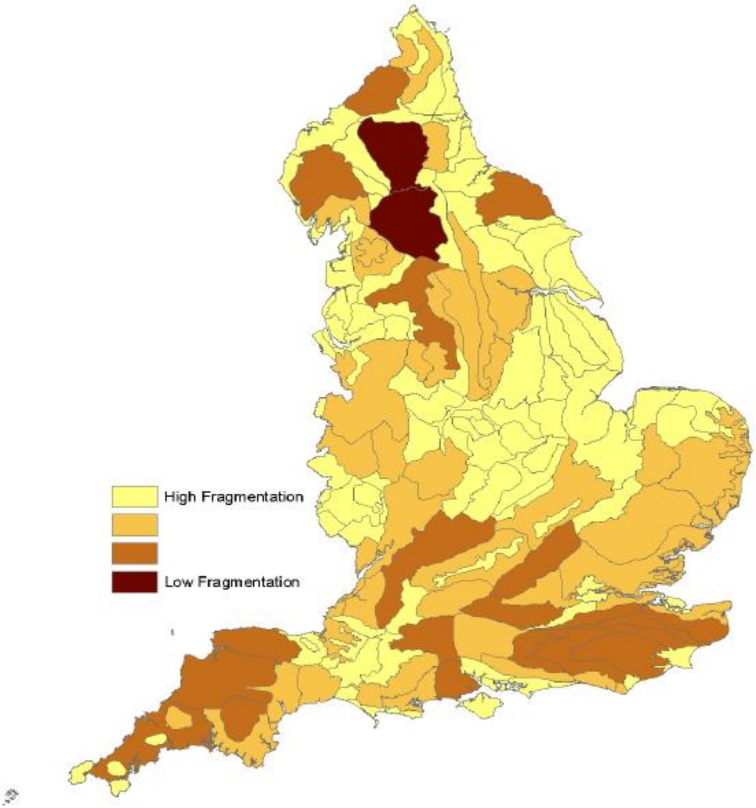
Agri-environment Schemes

Agri-environment schemes, supported through the CAP and co-financed by national funds, are the main national policy mechanism for environmental management of rural areas. They consist of voluntary agreements that provide annual payments to farmers and landowners to manage land in ways that limit the detrimental effects of agricultural activity.

Agri-environment schemes provide funding for the management of areas with statutory conservation designation, such as SSSIs and other wildlife sites that are not owned by central government or its agencies. These payments are no more than income forgone, such as reductions in crop yield, plus the additional costs of undertaking environmental management measures, for example, fulfilling requirements for the management of landscape features, such as hedgerows.

Different versions of agri-environment schemes have been used as policy tools in the UK since the mid 1980s and there are now more than 58,000 agri-environment scheme agreements, covering over 6 million hectares or approximately 66% of the

Figure 18 Levels of Habitat across National Character Areas.[227]



This analysis takes account of habitat extent and permeability land between habitat patches to produce a ranking from areas where habitats are most fragmented (lighter) to less fragmented and more connected (darker) agricultural land in England at a cost of £400 million a year paid by the UK government and European Union funds. Agri-environment schemes in England have recently been extensively reviewed and evaluated by Natural England, the key body for their delivery in England.

The current agri-environment schemes in England are the English Woodland Grant Scheme and Environmental Stewardship, which is further subdivided into Entry Level Stewardship (ELS), Higher Level Stewardship (HLS), Organic Entry Level Stewardship (OELS), Organic Higher Level Stewardship and an Uplands Entry Level Stewardship (UELS). The objectives of the Environmental Stewardship scheme include:

- protecting and enhancing habitats and species, landscape character and quality, the historic environment, soils and natural resources
- supporting the adaptation of the natural environment to climate change
- contributing to mitigating climate change, reducing flood risk and conserving genetic resources.

Most ELS agreements are dominated by three groups of management options: boundary options, mostly hedgerow management; low input grassland options; and management plans. The scheme has not been as successful as intended at delivering conservation of biodiversity, although benefits for some options have been demonstrated, and Natural England is seeking to improve the advice available to land managers. Options with the highest biodiversity benefits, such as skylark plots, wildflower meadows and overwintered stubble have had a relatively low take up.

The HLS agreements offer greater scope for reducing land use intensity, with individual agreements for ten years designed with the input of professional advisers. However, the option choice available to land managers depends on the existing environmental features present. The HLS offers the scope to maintain and restore degraded habitats as a range of habitat creation options, with a view to increasing the resilience of existing habitat patches by creating linking habitat. The scheme can act as a means of ensuring integrated management across an area to halt degradation of ecosystems (Box 18), but is resource intensive. Agreements require ongoing advice to land managers and would be expensive to apply at the landscape scale.

"Campaign for the Farmed Environment"

The "Campaign for the Farmed Environment" is a voluntary scheme that seeks to provide advice to maintain the environmental benefits provided by farmland, particularly resource protection (including water resources), farmland birds and farm wildlife, with management measures funded through the Entry Level Stewardship (ELS) but also with additional voluntary measures.

Training is being provided for agronomists, who are providing advice on the most effective measures for specific farms to farmers signing up to the campaign, in the hope that ELS agreements contain the right options or that farmers will undertake additional voluntary measures in the right places to create important habitats for wildlife and offer resource protection.

The campaign is seeking to retain uncropped land, but the emphasis is on the quality of outcomes rather than quantity and at least a third of uncropped land should support habitats for birds, insects and mammals. The campaign was formed in response to the loss of the environmental benefits previously provided by the set-aside scheme, which paid farmers to take land out of production, and will be replaced by a regulatory scheme if desired outcomes are not achieved.

Agri-environment Schemes and Biodiversity Loss

The expansion and intensification of agricultural land is recognised as one of the main drivers of environmental change through the transformation of natural habitats and land use change (Chapter 2, Annex A).[228] There is an extensive body of literature on the impacts of farming on biodiversity and about how farming practices could be modified to mitigate effects on biodiversity. Agri-environment schemes have been effective in delivering some biodiversity targets, such as that for 95% of SSSIs to be in favourable condition. Furthermore, 84% (about a million hectares) of the area of habitats identified as a national priority for protection and restoration are eligible for agri-environment schemes in one (Biodiversity Action Plan priority habitat).

In particular, the HLS and its predecessor schemes have been successful in restoring populations of farmland birds and other species,[229] and habitats of conservation importance such as hay meadows and calcareous grassland.[230] Research and evaluation programmes have allowed the schemes to be progressively refined. Although agri-environment schemes play a significant role in delivering biodiversity benefits, they have not halted continued significant losses in biodiversity in the UK. The overall scale of agri-environment schemes does not appear to have been sufficient to offset the pressures faced by the populations of species that continue to decline (Chapter 2).

Box 19 Examples of Integrated Management through Higher Level Stewardship

European heaths and shrublands are semi-natural ecosystems that have been co-evolving with human societies for millennia and are key habitats for species and landscapes valued by humans. These must be actively managed to maintain the habitat.[231] In Dorset, an Urban Heaths Grazing Partnership was originally formed in September 2005 by local authorities (Poole, Bournemouth, East Dorset, Christchurch, and Dorset County Council), nature conservation NGOs (Royal Society for the Protection of Birds, the Dorset Wildlife Trust and the Herpetological Conservation Trust), Natural England, Dorset Police and Dorset Fire and Rescue Service, working together to address management problems on Dorset's urban heaths. The urban heaths in Dorset, comprising nearly 2000ha of internationally designated heathland, are extremely fragmented and surrounded by the conurbations of Poole and Bournemouth. The partnership is working to change public perception of the heaths and their management, and aims specifically to:

- improve partners' abilities to prevent and tackle heathland fires and other undesirable incidents
- implement an area-wide education programme seeking to prevent abuse of heathlands and disturbance to its wildlife
- deliver a community action programme to build local support for heathlands
- provide integrated communication between all partners
- demonstrate effectiveness of actions.

Research has shown links between adverse effects on the heathland and the proximity of built development. The diverse effects that people and urban living have on the heaths have become known as 'urban pressures' and are now well understood and documented. These include fire setting, damage and disturbance to habitats and species by visitors, motor vehicle trespass, erosion, predation by domestic pets and urban-adapted predators such as foxes and crows, soil enrichment from dog excrement and fly tipping (including garden waste) and opposition by visitors and local communities to management measures such as tree felling and the introduction of grazing animals.

There is a need to restore habitats degraded by lack of management and resources by bringing back traditional management practices and removing encroaching scrub and trees. Active management of the heaths is critical to ensuring the survival of species of high biodiversity value, such as Dartford warblers, nightjars and the full range of British snakes, lizards and amphibians. A significant proportion of the urban heaths is now in seven HLS agreements, started in 2007, which will fund activities such as fencing, water supply, tree and non-native species removal and scrub and bracken control, and will allow integrated management across the sites.

The Pevensey Levels cover 4,300ha of marshland between Eastbourne and Bexhill-on-Sea in East Sussex. They are designated as a Ramsar site and SSSI because of their outstanding importance for birds, plants and invertebrates, which thrive in a mosaic of dykes, ponds and grasslands, with 184ha of the area designated a National Nature Reserve (NNR). Twenty years ago, there had been a significant loss of the wet grassland and the associated breeding bird numbers were in chronic decline. Through agri-environment schemes, the management of the whole area was improved by restoring water levels and improving cattle grazing, with the funding now provided through the Higher Level Scheme. The agri-environment scheme funding has had a multitude of benefits, restoring an important wildlife habitat and allowing associated businesses to flourish.

Agri-environment Schemes and Ecosystem Services

The recent review of ecological networks for Defra recommended that the government should promote economic approaches that will favour conservation management. This should be achieved by stimulating the creation of new markets and payment for ecosystem services, to ensure that the values of wider ranges of ecosystem services are taken into account in decisions that affect the management and use of the natural environment.[57]

The next generation of agri-environment schemes will aim to deliver multiple ecosystem service benefits, and this can already be achieved on a small scale but to do so at the spatial scale of a landscape will be challenging. The delivery of multiple ecosystem service benefits will also require a sophisticated level of knowledge about changes in the ecosystem services in response to land management measures.

Ecosystem services are usually dependent on processes that interact across a range of spatial scales and may require co-ordinated uptake of management interventions across a large area to be effective, such as flood management. This includes co-ordinated action across landscapes at a scale of intervention significantly higher than under existing agri-environment schemes.

Most existing agri-environment schemes make payment on the basis of proxies, such as the size of area under a particular management regime, rather than results. Future payments could be based on service delivery, but this is problematic for those services where provision is difficult to observe at the farm scale. Some services, where the link between management and service output can be identified more clearly and that are not dependent on collective action across a landscape, such as carbon sequestration or an increase in the number of target species, could be initial candidates for output based reward mechanisms. [232]

However, studies have suggested that existing agri-environment scheme measures can, to varying degrees, enhance the delivery of some ecosystem services benefits. These include:

- some measures, such as sowing wildflowers or pollen and nectar mix as arable field margins, that can increase pollinator abundance thereby supporting local pollination services for crops (Box 2);

- other measures that increase insect abundance through supporting natural pest control by insects;
- measures that provide flood alleviation (Box 15) and coastal protection supporting water regulation services;
- measures to enhance carbon storage in soil and vegetation, supporting carbon regulation services;
- measures to reduce soil erosion and to improve soil quality, supporting nutrient cycling services; and,
- measures such landscape conservation, cultural heritage, education and areas for recreational activities, supporting cultural services.

However, levels of ecosystem service provision are rarely determined by agricultural or forestry management alone. There is research ongoing to determine the land management measures necessary to deliver specific benefits, and to develop ways of targeting agri-environment schemes effectively to deliver landscape scale objectives. Natural England is running three ecosystem service pilots in upland areas (South Pennines, Exmoor, Dartmoor and Bassenthwaite) to understand what an ecosystem service approach will entail from a delivery perspective. The three Natural England ecosystem management pilots will seek to pioneer new approaches to paying for delivery of bundles of ecosystem services and will begin to identify the quantities and patterns of uptake of land management measures required in a locality to deliver the desired level and distribution of ecosystem services.

At the landscape scale, there is debate on whether intensive agriculture on a limited area is a more resilient system than less intensive agriculture across a larger area. In general, it appears landowners prefer to compartmentalise agricultural and environmental activities, undertaking measures such as field margins in a set area and maximising food provisioning services in other areas. The data available indicate that for arable systems it is more effective to separate activities than to attempt to undertake them together, with high quality habitat on small scale delivering more benefits than a lower quality habitat over a larger area. Habitat compartmentalisation along these lines appears to work well, but it is less certain how much habitat is required to support the full array of services required in any area.

Funding for Ecosystem Service Provision

Alteration of agri-environment schemes to deliver the explicit objective of a range of ecosystem services will be possible only if the specific limitations on the activities that can be funded through EU co-financed rural development programmes are addressed in the current round of CAP negotiations. To be delivered on the required scale, there would need to be agreement for more money from Pillar I of CAP, (the funding mechanism for the single farm payment), to be shifted to Pillar II (modulation), the funding mechanism for rural development programmes.

Creating any scheme for payments to maintain the delivery of ecosystem service benefits also requires an overarching vision for ecosystem service provision to be set out to act as a strategic framework. There is scope in many situations to achieve multiple benefits at the same time – joining up farming, management of flood risk and nature conservation. However, achieving this will require linking the funding streams that support these hitherto different policy areas and co-ordination with other conservation policies, such Ecological Restoration Zones.

There is already an acceptance by stakeholders, such as the National Farmers Union (NFU), that land should be managed for services other than solely food or timber production, as land managers support many services for which they are not remunerated. There have been calls for the separate ecosystem service initiatives of Defra, the Environment Agency and Natural England to be joined up, and for greater clarity on management measures or instruments to deliver policy and on the timescales for implementation.

Such a strategic framework would also create possibilities for alternative funding sources, such as emerging carbon markets or water companies, although it is not clear if true markets for ecosystem services would develop. One option would be through the introduction of reverse auctions, where land managers would bid to provide defined ecosystem service benefits against other land managers, but it is debatable whether this would create a true market, [233] and it may compete with any system of biodiversity banking introduced.

Research at the National Centre for Ecological Analysis and Synthesis (NCEAS) in the USA took an ecosystem and derived a value for the ecosystem services per unit area, using existing data, expressed as a dollar value per unit for each ecosystem service.[234] Unintended consequences of such an approach could be that damaging activities become concentrated in the areas that support low value ecosystem services. These may be ecologically important, despite not directly supporting human benefits, thereby damaging services disproportionately. Equally, there might be increased competition for areas that support high value services, raising costs of key ecosystem goods and services, such as food.

There would also need to be inclusion of the views of stakeholders other than just land managers (Section 4.11). For example, as part of a RELU funded project, farmers have expressed a willingness to change land use practices to provide services that are benefit the public at large. Examples include controlling/retaining run-off from farm land and storage of flood water on farm land in flood plains, including that previously protected from flooding by embankments. [235] As shown by debates over coastal retreat projects, such changes are likely to be contentious. However, increased numbers of demonstration projects may show communities how public benefits can be realised from changes in land use and could inform a change in attitudes and raise familiarity with such schemes.

Offsetting and Ecosystem Services

At present, there is no system in place to trade credits or to operate habitat mitigation, offset or conservation banks in the UK. There is considerable policy interest in this area, and both Defra and the European Commission DG Environment have commissioned recent reports on the options for implementing such policy frameworks. Such approaches are also recommended by the recent TEEB report.

Offsetting would be implemented to provide additional biodiversity benefits rather than to deliver conservation outcomes already required under legislation, such as the Habitats or Birds Directives. The Defra and IEEP reports suggest that for widespread biodiversity that does not have statutory protection, a system of payment in lieu of credit could be economically possible, as there would be sufficient volume to reduce the transaction costs of trading and it would deliver biodiversity gains. In the USA, endangered species banking (conservation banks) are estimated to generate $370 million gross revenues a year and wetland credit banking approximately $1 billion a year.[236]

The aim of such regulatory frameworks is to offset environmentally damaging activities by purchasing credits from actions with beneficial biodiversity outcomes. Most offsetting policies implemented focus on biodiversity rather than ecosystem services. Credits are not usually linked to the debit – the damaging activity – that they are purchased to cover. The intention is that there should be no net loss, or preferably a net gain, of biodiversity in terms of species composition, habitat structure and ecosystem service benefits. The credits compensate for the residual damage, which is the damage done to biodiversity after all possible mitigation measures have been implemented at the affected site. After an assessment of biodiversity likely to be affected and the extent to which on-site mitigation measures could reduce impacts, the amount of credits required is calculated to ensure there is no overall net loss of biodiversity.

However, the unit value of credit provision for biodiversity needs to be location-specific to be relevant to habitats and biological species. For example, “lowland meadow credits” could be generated only on land with ecological potential to support them. The value of biodiversity can also vary between areas, with a habitat or species being far more common in one area than another. The relevant trading scale will depend on the habitat and species. For example, inter-tidal saltmarsh credits could be traded within the southern North Sea (between eastern England and the Netherlands) whereas chalk grassland would be limited to counties in southern and eastern England.

An offsetting system could incentivise landowners and land managers to protect, recreate or restore habitats to supply credits. The market created by offsets could be similar to that of tradable permits for carbon emissions (POSTnote 354). Potentially there could be some major ecological benefits from offsetting, depending on how schemes were operated, including:

- addressing the cumulative impacts of many small scale land use changes
- creating funds to implement green infrastructure strategies, such as river restoration projects, or creation of seasonally flooded areas for flood storage
- strategic land management measures to increase resilience, such as pooling offsets for ecological networks to reduce habitat fragmentation across landscapes and to implement buffer zones around important areas of semi-natural habitat.

Issues Arising from Offsetting

It has been suggested that an offsetting system could operate parallel to credits for different ecosystem services for a single parcel of land, with the management measures of one type of credit not being allowed to conflict with another. However, the EU is clear that any system should compensate for damage to biodiversity rather than be used to maintain ecosystem service provision, as there is potential for conflict between the two.

The most valuable locations for some ecosystem services may not coincide with those that support the most biodiversity. For example, an assessment of the distribution of biodiversity, carbon storage, recreational use and agricultural value within Great Britain showed that habitats that are important carbon stores have relatively few species of conservation concern and suggested that land use policies that encourage more carbon storage were unlikely to deliver national scale biodiversity commitments. There is a much stronger correlation between those areas important for agriculture and for biodiversity, but biodiversity outcomes are likely to be traded off against agricultural productivity in such areas.[237]

In areas where the resilience of ecosystems has been substantially reduced through extensive habitat fragmentation, and where there are low levels of biodiversity, further loss could lead to environmental limits being exceeded. Most biodiversity offsetting or banking schemes that operate in other countries such as the USA and Australia have an "upper threshold" limit beyond which offsetting or banking is not appropriate. For example, rare biodiversity in an area is considered to be non-substitutable.[238] Where significant impacts on ecosystem services cannot be compensated for using known or proven techniques, offsetting or mitigation is not permitted (POSTnote 369).[238]

Using biodiversity offsetting to ensure that suitable habitat mosaics are retained within landscapes would also be heavily dependent on the availability of local expert judgement, and there needs to be a far better evidence base on what is needed, and where, within landscapes before this would be viable at a large scale to enhance resilience of ecosystem service provision.

For example, landscape habitat mosaics should be managed to provide for the different habitat needs of the guilds of pollinators to provide resilience in pollination services (Box 2). A guild is a group of species that exploit the same resource, in this case flowering plants, in a similar way, with different guilds being classified according to how they acquire nutrients, their mobility and ways of feeding. A range of different pollinator guilds with differing lifecycles and physiological responses (Section 2.5) should provide resilience to inter-annual variation in climate or other abnormal events, such as invasive species or disease. However, not enough is understood about the differing habitat requirements of guilds of pollinators across landscapes to achieve this.

The recent review of ecological networks for Defra recommended that if a formal system of biodiversity offsets is to be introduced, pilot schemes should be established to test and refine its operation. In particular, the evidence base needs to be further developed to:
- refine creation and restoration techniques for certain habitats;
- establish the appropriate multipliers needed to ensure full compensation; and,
- to develop rules for offsetting 'out of kind' (where damage to one type of habitat is compensated for by providing another).

Other possible conflicts that could arise include:
- The impact on local communities. The compensating habitat provided elsewhere may not provide benefits to the community in the immediate locality of the original site.
- The risk of becoming a "licence to trash" if the offsetting scheme makes compensation measures for habitat loss easier and cheaper, or land use changes are permitted that otherwise would have been refused, due to the level of impact.
- The system offsets not being underpinned by a clear set of principles, leading to a reduction in the protection afforded to existing semi-natural habitats and the undermining of the establishment of ecological networks.
- That the most suitable sites for habitat banking could be sold as offsets without management measures to enhance ecosystem services being implemented, unless the regulator requires it.
- That some landowners may actively seek to degrade high quality habitats to qualify for credits to improve them, which could also lead to conflicts with the aims of agri-environment schemes. Prior to the implementation of a habitat banking scheme there would also be a strong incentive for landowners and developers to degrade sites before becoming subject to the restrictions, thereby reducing compensation costs.
- If regulation and enforcement is poor, that the system could be corrupted by virtual rather than real biodiversity offsets, especially if credits are resold multiple times.

Property Rights

Property rights over natural resource systems define rights and responsibilities in the use or conservation of environmental goods and services. These rights may be held by a number of different parties, such as those held by a farmer under an agricultural tenancy, and those held by the public to access land under Public Rights of Way Regulation.

Stakeholders extract value from the use of ecosystem services and exercise influence through property rights and entitlements.

Property rights confer entitlements to the flow of benefits from natural capital, usually associated with its direct use, such as through agriculture, fishing or timber extraction. The ecosystem approach requires explicit links to be made between flows of ecosystem service benefits and stakeholder values, but property rights are not yet defined in relation to the full range of ecosystem services and their public benefits (Table 15). For example, the value of land is usually measured in terms of the profits derived from agricultural use and outputs, as property rights usually give precedence to provisioning ecosystem services.

There is an array of institutions and rules that govern the relationship between stakeholders and natural capital use, such as CAP or licenses for fishing. Property rights can relate to areas of land and their uses or to individual biological species and usually attach to direct uses, such as shooting rights for game species, but can also include indirect uses, such as requirements to protect endangered species. There are two key governance aspects of property rights and ecosystem services:

- entitlement to the benefits of ecosystem services conferred by governance institutions to allow stakeholders to extract value; and,
- the applicable regulatory regime for the use of benefits (whether private, state or common).

It should be noted that entitlement in property rights to benefit from natural capital derives from societal preferences that have progressively shifted from acceptance of feudal ownership to increasing imposition of responsibilities to maintain public benefits. Various planning acts have sought to restrict changes in land use to comply with planning policy developed at national and local level.[239] Other acts, such as the National Parks and Access to the Countryside Act 1949, and the Wildlife and Countryside Act 1981, introduced designations such as National Parks and Sites of Special Scientific Interest and restrictions on the rights of private landowners to institute land use change. The transition in emphasis from private gain to the safeguarding of public benefits valued by society is reflected in the creation of agri-environment schemes.[239]

There is the potential for future legal challenges on the basis of public interests being damaged by management actions to enhance private benefits, such as suing for damages for loss of benefits or injunctions against actions that may result in their loss. An example is the case of the scallop dredging fishery in Lyme Bay, which had a damaging effect on protected marine biodiversity on rocky reef habitat, such as pink seafans. The right to fish was curtailed by a the UK Government, which closed a 60 square nautical mile area in July 2008 using a Statutory Instrument to provide protection against damage caused by bottom-towed gear, protecting public benefits from exploitation by private interests.[239] Lyme Bay was designated an SAC in August 2010, conferring further legal protection.

Ensuring resilient ecosystem service provision within environmental limits, now and in the future, raises a number of other legal implications:

- How to design property rights and related institutions to capture fully the properties of ecosystems and the full range of ecosystem services they relate to, given the uncertainties regarding the functioning of ecosystems. This would include consideration of acceptable limits for provision of ecosystem services.
- The need for clear, enforceable and transferable property rights, for markets for environmental goods and services to function, Property rights that apply to natural resources should cover the full range of ecosystem service benefits to clarify which parties can claim and control rights of use.
- Ensuring the equitable distribution of benefits to avoid the legitimacy of the system of property rights being undermined by legal challenges or lack of stakeholder acceptance
- Bridging the gap between public benefits and private gain, where conflict arises between stakeholder interests in respect of private gain from the flow of a specific ecosystem service benefits versus loss of a wider array of public benefits. For example, in areas of significant water stress, it may be necessary to restrict or revoke water abstraction licences to protect water supplies and the ecology of rivers.

There is also the potential for conflict between property right holders that observe good stewardship and others that benefit from this without contributing. At the landscape level there can be a ‘freerider effect’, where responsible landowners are effectively penalised for good stewardship by those who profit from measures taken to support ecosystem service benefits, while taking no measures on their own land. This can be addressed through regulation to ensure that burdens are shared equally. For example, farmers and agro-forestry users in Switzerland are grouped in joint agri-environment schemes, where all individual participants have to meet their own targets to collectively trigger payments.

Table 5: Public and Private Benefits from Ecosystem Services and Their Temporal and Spatial Scale[239]

	MA Ecosystem service	Potential benefits derived	Scale of benefit
Provisioning services	Fresh water	Public supply, industry and irrigation	Catchment scale
	Food (eg crops, fruit, fish, etc)	Private crop production and low intensity cropping (public or private)	Local scale, seasonal
	Fibre and fuel (eg timber, wool, etc)	Private or public harvesting, grazing, building materials, etc	Local scale, seasonal
	Genetic resources (used for crop/stock breeding and biotechnology)	Private or public harvesting or breeding of genetic resources	Potential wide-scale and long-lasting benefits
	Biochemicals, natural medicines, pharmaceuticals	Private or public harvesting or breeding of resources	Potential wide-scale and long-lasting benefits
	Ornamental resources (eg shells, flowers, etc)	Private or public harvesting	Local, but potentially long-lasting
Regulatory services	Air quality regulation	Public benefits to air quality and health	Local to medium-range
	Climate regulation temperature/precipitation, GHG sequestration, etc)	Public benefits arising from climate stability	Local and global benefits, (local long-lasting
	Water regulation (timing scale of run-off, flooding, etc)	Public benefits from stable flows and flood attenuation	Catchment scale, and long lasting
	Natural hazard regulation long (ie storm protection)	Public benefits from mitigating extreme events	Localised beneficiaries, lasting
	Pest regulation	Public and private benefits from natural pest regulation	Localised and enduring
	Disease regulation	Public and private benefits from natural disease regulation	Local or medium-range and enduring
	Erosion regulation	Public and private benefits of soil conservation and reduced siltation of waterways contributing to declining biodiversity	Catchment-scale impacts that may be long lasting
	Water purification and waste treatment	Public benefits through absorption of waste materials and improved quality of water supplies	Catchment-scale impacts that may be long lasting
	Pollination	Public and private benefits from natural pollination services	Localised and enduring
Cultural services	Cultural heritage	Public benefits from maintaining culturally important sites	Localised or medium range and enduring
	Recreation and tourism	Public benefits from amenity and private benefits from profit from tourism and recreation activities	Localised to medium-range and enduring
	Aesthetic value	Public and private benefits provided by landscape	Local, medium range and potentially global, enduring
	Spiritual and religious value	Public benefits supported by landscape functions	Local, medium range and potentially global, enduring
	Inspiration of art, folklore, architecture, etc	Public benefits supported by landscape functions	Local, medium range and potentially global, enduring
	Social relations (eg fishing, grazing or cropping communities)	Public benefits through habitat support of local communities	Local to medium range, enduring
Supporting services	Soil formation	Public benefits through creation of fertile soil	Catchment-scale, long lasting
	Primary production	Public benefits through productivity of ecosystems	Local, medium range and potentially global, enduring
	Nutrient cycling	Public and private benefits through maintenance of productive cycles, fertilising soils and metabolising potential pollutant nutrients	Catchment-scale, long lasting
	Water recycling	Public and private benefits through renewal of water systems	Catchment-scale, long lasting
	Photosynthesis (production of atmospheric oxygen)	Public benefits through oxygen production and carbon dioxide sequestration	Local scale, long lasting
	Provision of habitat	Public and private benefits through maintenance of characteristic biodiversity	Local, medium range and potentially global, enduring

These schemes often operate at the river catchment level and expert advice (through an environmental audit) helps to identify the opportunities for conservation and environmental management.

Payments for Ecosystem Services

Deriving a monetary value for benefits of ecosystems makes explicit the need to pay for the provision of services and to protect them. There is considerable policy interest in the development of markets for ecosystems, as exemplified by the development of carbon trading schemes (POSTnote 354), through tools such as payment for ecosystem services (PES). It is argued that by linking up beneficiaries and providers, PES approaches can strengthen integration between the natural environment, the economy and society and engage a broader spectrum of stakeholders.[240]

As land managers rarely receive income for carbon storage, water regulation, maintenance of air quality or protection against natural hazards, they have little incentive to conserve or manage ecosystems to maintain these services. In general, realising marketable commodities, often through the modification, simplification and degradation of ecosystems, will take precedence. The aim of PES is to protect ecosystem services by providing an economic incentive to land managers to adopt land use or management practices favourable to the protection of ecosystem services to help to realign the resulting private and public benefits.

PES have been defined as a 'payment for an ecosystem service benefit as a voluntary transaction where a well-defined ecosystem service is bought by at least one buyer from at least one supplier, the payment being conditional on the benefit being delivered'.[241] The way PES operates depends on the numbers that benefit from the service and whether the supply and demand for it are located in a distinct geographic area. In cases where small groups benefit in a specific location and the supply arises in a specific location, then arranging such transactions is more straightforward.

The PES approach of paying subsidies not to undertake polluting activities could be construed as rewarding parties for adopting management practices they should undertake as part of good practice or stewardship, conflicting with the "Polluter Pays" regulatory principle. This is intended to reflect the value of natural resources within public and private decision-making and to bring private incentives in line with society's interest, for example, through taxes on polluting activities.

Regulating ecosystem services that provide global or regional benefits, such as climate regulation or pollination, are public goods and there are unlikely to be a clear private beneficiaries. If the supply of the benefit cannot be assured through regulation, then landowners may need to be paid by governments or intergovernmental organisations. International examples of regulatory authorities paying landowners on behalf of beneficiaries include well known examples such as forest maintenance in Costa Rica or catchment protection measures in the Catskill mountains in the USA. Similarly, in the UK, Defra has classified agri-environment schemes and SCaMP (Box 14) as PES.

However, none of these payment schemes for ecosystem services by regulatory authorities fulfils the technical definition of PES or creates economic markets for ecosystem services. An example that does fall within the technical definition of PES is the protection of the aquifer for Vittel mineral water at the foot of the Vosges Mountains in north-eastern France (Box 19). There are a number of challenges to implementing a market for payments for ecosystems services that would fall within the full technical definition, including:

- A lack of understanding of how given management measures will affect specific ecosystem processes and translate into changes in ecosystem service output.
- Whether payment should be on the input of expenditure to maintain natural capital assets or for the output of ecosystem service flowing from natural capital assets.
- The appropriate spatial scale for PES, both the scale over which the ecosystem service is delivered as well as the spatial scale of beneficiaries.
- How to package multiple ecosystem services into PES schemes.
- How the delivery of ecosystem services can be measured and monitored.
- The risk that paying farmers to provide the broad range of ecosystem services could lead to higher market prices for goods from provisioning services, leading to increased land use change and loss of ecosystem service provision in other locations.
- Ensuring that PES delivers benefits beyond what is already required by existing incentives and regulation.

Box 20: The Vittel Payments for Ecosystem Services[242]

To be labelled 'Vittel', the water cannot contain more than 4.5mg/l of nitrates, nor have traces of pesticides and must meet other French regulatory standards for mineral water. In the 1980s the intensification of agriculture, moving away from hay-based cattle ranching systems to maize-based systems, led to increased risks of nitrate and pesticides in the aquifer. Because of the configuration of the sub-catchment (all the relevant farmlands were located upstream from the spring area and varied in the percentage of land within the relevant zone that influenced the spring waters and their distance to the spring), each farmer could individually influence the nitrate rate. Extensive hydro-geological modelling was conducted in the area and showed that ensuring a nitrate rate of 4.5mg/l in "Grande Source" required maintaining levels of nitrate levels at 10mg/l up to 1.5 metres below the surface. The sub-catchment was modelled at sub-catchment, farm, and plot level to test the technical and economic feasibility of the proposed alternatives, which were then tested on farms. Several practices were identified to maintain this nitrate level and zero pesticides level:

- give up maize cultivation for animal feed (land under maize production showed nitrates rates of up to 200mg/l in the root zone
- adopt extensive cattle ranching including pasture management (hay and alfalfa rotation so that farms produce all animal feeds themselves)
- reduce carrying capacity to a maximum of one cattle head per hectare (equivalent to one cow of 600kg in weight)
- halt use of all agrochemicals (fertilisers being substituted by manure and no pesticides)
- compost animal waste and apply an optimal amount in fields
- balance animal rations to reach optimal milk productivity and farm profitability
- modernise farm buildings for optimal waste management and storing.

However, it should be noted that a precise link between farm practices and nitrate and pesticides in the aquifer was not established nor the contribution of individual farms to water quality in the spring. Once the science to inform the management measures required was available, it took a further ten years to negotiate with and convince all the relevant farmers to participate in the scheme. One of the main negotiation issues was whether payments should be based on the economic benefits gained by the company owning the Vittel brand or on the income forgone by farmers (opportunity costs). Eventually an incentive package to undertake the measures was agreed that included:

- long term security through 18 or 30 year contracts
- abolition of debt linked to land acquisition
- a subsidy of, on average, about €200/ha/year over five years
- up to €150,000 per farm to cover the cost of all new farm equipment and building modernisation
- free labour to apply the composted animal manures to farmers' fields, ensuring the optimal amount was applied to each plot
- free technical assistance including annual individual farm plans and introduction to new social and professional networks.

By 2004, 92% of sub-catchment was protected with all 26 farms in the area had adopting the new farming system and 1,700ha of maize no longer grown. A number of insights can be drawn from this example, not least that establishing PES programmes is a complex undertaking, the active participation of stakeholders is required to address the social, legal, political and communication issues involved, and that a business case can be made for the participation of the private sector in payment for ecosystem services.

- The appropriate governance structures to resolve the lack of clarity as to who should pay for ecosystem service benefits that are essentially public goods. Within national spheres governments can play a role, but services that have global benefits require institutions to broker deals between suppliers of ecosystem services on an international basis.

4.11 Adaptive Management

Some commentators have suggested that ensuring resilience of ecosystem service provision is probably best considered as part of the process of adaptive ecosystem management.[182] Management of natural resource systems has been based upon the assumption that the behaviour of complex ecological systems in response to pressure can be predicted and that science will reduce uncertainties in relation to management options. [243] Current management strategies attempt to produce stabilised resource flows (Section 5.2), by suppressing disturbance, such as fires or floods, and reducing diversity within ecosystems, with impacts on slowly changing regulatory ecosystem variables, such as soils.[244] These approaches and the assumptions underlying them have led to a loss of resilience in natural resource systems and an increase in the risk of environmental limits being exceeded (section 5.1).[245] The increased vulnerability of SES to environmental change may cause social disruption through impacts as such as reductions in food security and may lead to long term declines in the productivity of the natural resource systems being managed. [246]

By contrast, the adaptive management approach, assumes that abnormal events are inevitable, knowledge of systems and their interactions will always be incomplete and that human interactions with ecosystems will always be evolving.[247] The approach acknowledges that the natural resource being managed will always change, so management of human activities must adjust and conform to these changes, including unexpected events.[248] Adaptive management requires uncertainties to be identified and then to 'test' possible management measures to see if they help to achieve desired levels of ecosystem service benefits. This requires management strategies to test different management actions while monitoring outcomes and accumulating knowledge to feedback as corrections to them as understanding of the dynamics of the system increases.[244]

Management actions should actively seek to identify the assumptions underpinning management measures, such as that flood management measures, will always have beneficial effects on wetlands (Box 15). Management actions are intended either to avoid thresholds being crossed through increasing the resilience of ecosystems or to allow the ecosystem to recover following disturbance.

A key challenge for such management is to develop institutional structures that match ecological and social processes operating at different spatial and temporal scales and to address the linkages between those scales (Chapter 4). Multi-level governance of complex ecosystems, faced with ecological and social change, needs constant adjustment, requiring flexible institutional arrangements.[249] Such arrangements are difficult to implement, as they are 'messy', non-hierarchical and require ways of ensuring local organisations interact with each other and with other organisations at different levels. Referred to as "adaptive co-management", management arrangements rely on public participation (Section 4.9) across a diverse set of interest groups operating at different levels, from local users, to local government, to regional and national organisations and occasionally to international networks.[250]

Passive versus Active

A core feature of adaptive management is a conceptual or quantitative model of the natural resource system being managed. The model is not used to predict policy consequences, such as whether a decision will decrease or increase resilience, but to increase understanding of the system behaviour and the assumptions that can be usefully evaluated through changes in management.[251] There are two basic kinds of adaptive management, active and passive.

Active adaptive management is a substantially riskier approach, but can be considerably more informative in determining where ecological thresholds lie.[251] Active adaptive management actions are deliberately taken to 'test' the natural resource system to understand how it behaves. A set of competing hypotheses about what causes shifts in ecosystem states is tested through a structured set of management measures to reveal key variables, such as slow changing ones in soils, as well as system potential through a comprehensive monitoring programme. This should create a diversity of management options for responding to uncertainty and unforeseen ecological shocks.[244]

Passive adaptive management consists simply of using whatever information arises from management actions to improve understanding of the system. In general, passive adaptive management to increase the resilience of natural resource use is more publicly acceptable. For example, once a change in land use within a river catchment can be linked to a decline in water quality, regulators can work with stakeholders to put in place measures to mitigate the impacts through agri-environment schemes or to restore the original habitats and monitor whether the outcomes are successful.

Measures such as ecological restoration, ecological networks and changes to agri-environment schemes could all be implemented within active adaptive management frameworks. However, if, for example, an upland area in a river catchment were deliberately degraded through intensive agricultural use, to determine where the threshold for flood risk downstream occurs (Box 14), it would raise expense, ethical and legal concerns. In addition, most management measures for natural resource systems require public expenditure. Implementing measures with a known risk of failure or negative consequences for ecosystem service benefits provision would be politically challenging. In these cases, use of scenarios is an alternative for understanding key variables and increasing understanding among key stakeholders (Section 4.9).

Monitoring

Active adaptive management approaches require information about feedbacks from the impacts of decisions taken and potential new risks involved in the interaction of changing social, economic and ecological conditions. Management decisions are regularly revisited and changed as knowledge advances, guided by monitoring to keep track of environmental change. If a decline in an indicator that indicates a degradation in the resilience of ecosystem service benefit provision such as functional diversity, is detected, new intervention strategies would be triggered to reduce the drivers of change in the ecosystem.

Rather than being an inherent characteristic of ecosystems, resilience is a dynamic response of key variables in the ecosystem to a particular set of circumstances. The key requirement is not to determine the type of ecosystem structures responsible for a particular process, but to define how resilient a given ecosystem state is. Resilience, if described as the inverse of recovery time, is relatively easily to measure, if the state to which the system is recovering can be defined.

However, the science remains largely theoretical as, although these concepts can be modelled, large scale experiments to determine them quantitatively in different ecosystems are expensive and difficult. Some aspects of resistance to disturbance and the ability to recover from shocks have been studied in some ecosystems, such as lowland heath resilience under different management regimes.[252] However, the evidence base to determine how to enhance the resilience of most aspects of ecosystems is lacking, due to the paucity of studies that directly focus on resilience. [182]

In addition to modelling, the generation of data sets for natural resources such as land use cover change, changes in freshwater quantity and quality, stocks and flows of ecosystems services and trends in human uses of ecosystem services as well as indicators from these datasets to inform policy are needed at the global and national level. Indicators of gradual ecosystem change and early warning signals are desirable for adaptive management, but may be difficult to define (Annex B). Consideration of the wider aspects of SES, in terms of robustness, durability, stability and resilience (Figure 16) will be even more challenging.

The general capacity of most ecosystems to experience transient shocks, while retaining essentially the same function, structure, feedbacks and hence levels of ecosystem service benefits remains highly uncertain. The majority of data available relates to provisioning benefits (such as agriculture, forestry and fishing) with less known about supporting services (such as nutrient cycling) and regulating services (such as flood regulation) . Although various ecological attributes have been used as surrogate measures for ecosystem resilience, there are no universally accepted measures.[71]

Monitoring against Environmental Limits

Ensuring the resilience of the provision of ecosystem service benefits requires precautionary limits to be set, given the uncertainties in the understanding of ecological processes (Section 4.10). Existing research indicates that the range of potential indicators (Annex D) is diverse, but they can be grouped into those that relate to the ability of the system to resist change, or to the speed at which it recovers.[71]

With environmental limits, the critical variable is the output of ecosystem service benefits. Plausible adaptive management scenarios, based on the impacts of direct and indirect drivers of ecosystem changes on ecosystem service provision, would be a first step in limit and target setting for resilience using relevant indicators. It is unlikely to be possible to derive direct measures of *all* ecosystem service provision and it may be necessary to use facsimiles of, or proxies for, benefit provision.

For example, research is ongoing to establish measurements at the field scale to assess the effectiveness of agri-environment schemes. These will be a mixture of direct measures, such as soil carbon, and proxies such as the numbers of a particular species present, with expert panels to derive indicators for cultural services. The number of bumblebees present in an area can serve as a proxy for pollination services instead of direct measurement of actual pollination rates. There will be a need to validate such proxies used to determine the link to direct changes in ecosystem service provision. An example is the effect on grassland productivity when fertiliser input is reduced and replaced with nitrogen fixing legumes, such as red clover, and the impacts on other ecosystem services.

However, there are also issues of scale for developing ecosystem service indicators, as they can operate over widely different geographic and temporal scales. For example, soil carbon storage can be measured in a square metre of soil whereas flood protection services need to be assessed at the catchment level. A more tractable approach than identifying key aspects of biodiversity that maintain ecological processes, is to measure the quality of habitats that support relevant organisms, structures or processes. However, issues about the diversity of species present in a habitat become more critical when considering the resilience of ecosystem service benefit provision in response to environmental change.

6 Environmental Risks and Limits

Overview

- Environmental policy is, by necessity, becoming more sophisticated, and is no longer solely defined in terms of scientific and ethical concerns for the natural environment versus economic development. There are now additional concerns about which aspects of the natural environment need to be protected to ensure human well-being and future economic opportunities.
- Public attitudes affect the policy response to risks to well-being, and public acceptance may determine whether policies responding to a risk are implemented. There are significant constraints on communicating environmental risks to the public successfully.
- The consequences of abrupt ecosystem changes on a large scale could affect ecological security to such an extent that it is rational to minimise the risk of breaching ecosystem thresholds, given that these changes may be irreversible, even if there is uncertainty as to exactly where these lie.
- Vulnerability analysis can be used to determine possible causes of disruption in coupled SES, such as food systems, although the systems may be the result of complex interactions. Development decision pathways could be used to enhance the resilience of the system to possible disruption and to lessen the risk to ecological security.

6.1 Environmental Limits and Uncertainty

Due to the uncertainties involved, there has been relatively slow progress in defining ecological thresholds. Abrupt changes in natural resource systems can result from any discontinuity in the ecological processes underpinning ecosystem service provision and can occur at a range of scales from the local to the global (section 2.6). The ecological thresholds for such changes are subject to two types of uncertainty:

- 'epistemic' uncertainty, that is, arising from how well a natural resource system is understood, for example, where a threshold lies. This type of uncertainty can be reduced through increased understanding of the system.
- 'aleatory' (from the Latin word for dice)uncertainty, arising from the inherently unpredictable nature of the system. Although such uncertainty may be recognised (a 'known unknown'), it cannot be reduced through greater understanding.

However much is learnt about the component interactions of natural resources systems, their future behaviour and response to abnormal events will remain unpredictable. This scientific uncertainty increases the importance of societal values in relation to setting environmental limits, as different societal value sets lead to different weightings for where acceptable provision of ecosystem service benefits lies. Experts may be more able to make judgements about what the likely outcome of policy alternatives could be through ecosystem assessments and scenarios. However, identification of those outcomes considered desirable is a matter of societal choice, particularly when increases in one ecosystem service benefit are likely to be traded-off at the expense of others.

Scientific evidence can play a crucial role in informing the policy debate about maintaining natural capital, but the setting of environmental limits cannot be depoliticised or resolved solely through the collection and analysis of scientific evidence. In situations where uncertainty exists and there is a lack of social consensus on desirable outcomes, polarised debates may occur.

Decision-making cannot be based solely upon the science in these circumstances, as among other things, it may be disputed, regardless of the validity of research findings.[253] High uncertainty is likely to stymie the policy process in defining environmental limits, as different stakeholder groups will use the existence of uncertainty as a rationale for arguing for levels of natural resource use that favour their immediate needs.

Scientists also may hold a range of views. As the stages of dealing with risk (assessment, evaluation and management) all involve value judgements they can be influenced by such views. If scientists are perceived to be advocates for a particular agenda, the public is more likely to distrust the conclusions made from scientific data and misinterpret uncertainties.[254] The most recent version of UK government guidelines on the use of scientific and engineering advice in policy making states "the use of evidence is essential and scientists, engineers and policymakers must also ensure that they include evidence of any differing perspectives of risk as part of any decision making process".[255]

Science is unlikely to be able to resolve policy issues over environmental limits where significant uncertainties remain, although understanding these uncertainties can help to define possible future scenarios, as illustrated by the work of the Intergovernmental Panel on Climate Change (IPCC). Uncertainties are often interpreted as evidence that the scientific models explaining how natural resource systems function and environmental risks arising from changes are wrong. This masks the actual debate between opposing value sets of perceived unacceptable losses by one group and the perceived gains of another.

Uncertainties in scientific foresight about what will be the outcome of crossing ecological thresholds may result in scientific analysis being contested. To accommodate uncertainty, decision making strategies should consider a range of plausible alternative outcomes,[256] and choose management strategies for natural resource systems that are sufficiently robust across as many of these futures as possible (Box 20).

Values and Risk

Risk is a feature of all human impacts on natural resource systems that have effects that are more or less uncertain and that yield both a benefit and cost. The perception of risk involves a judgement as to the degree of uncertainty and the expected magnitude of loss or gain. Policymakers and stakeholders need to consider whether environmental risks can be identified, measured and managed and the point where the level of the risk relative to reward is unacceptable.

Risk-based environmental policies have generally involved identifying the most significant environmental hazards facing a geographical location or economic sector, estimating the probability of exposure to the hazard and the nature of resulting impacts, and assessing the most cost-effective means of reducing that risk to a level that is acceptable to society. This policy approach is driven by known risks that can be quantified with some certainty to justify the resultant public expenditure or regulatory burden.[257]

The policy response to risk is mediated by the public response to a risk, and public acceptance may determine whether policies to manage a risk are implemented. There are significant constraints on communicating environmental risks to the public successfully. Factors including psychological makeup, social norms, personal habits, structural conditions (the co-evolution of institutions and society with technology) and socio-demographic patterns will all condition how individuals respond to information about such environmental risks and whether they can be addressed by policymakers (POSTnote 347).

For example, research suggests that groups with different values will have widely differing views as to where the limits of acceptable environmental risk lie.[258] These profound differences in attitudes between individual value systems create difficulties engaging the public in responses to environmental risks in existing policy frameworks. Values translate into policy actions because they frame how societies develop rules and institutions to govern risk.[259]

The values that underpin decisions become more diverse and contradictory as one moves from the local and small scale up to the landscape, regional or national scale with multiple stakeholder groups.[260] Given the range of geographic scales over which ecosystem services occur, this could create significant problems in defining where environmental limits lie and attitudes to the risk of exceeding them.

Perceptions of Risk

Risk is generally regarded as a combination of the expected magnitude of loss or gain and the variability of that expected outcome. However, *perception* of risk works rather differently, with two important components: the fear factor of the potential outcome, and the control factor, that is, the extent to which human actors consider themselves in control of events. When risks combine both dread and lack of control, they are perceived as very great and hence lack acceptability.

Available evidence on human risk preferences suggests humans are risk-averse when considering potential gains in benefits, but will often take significant risks to avoid losses. For example, a land manager responding to an expected loss in ecosystem service benefits may be risk-averse to adapt actively to environmental change, thereby risking further losses.

In addition, whether humans perceive themselves as operating in the domain of gains or of losses depends on how a decision is framed, and the reference point used to judge these losses and gains. Ecological thresholds are not a suitable

Box 21 Thames Estuary 2100

Rising sea level, increasing development and ageing defences mean flood risk is increasing for London and the Thames Estuary. The TE2100 project was set up by the Environment Agency to develop a flood risk management plan for London and the Thames Estuary for the next 100 years. TE2100 aims to develop a plan with the uncertainties of climate change and different possible social and economic futures. The plan includes decision pathways to identify various options for future flood risk management and to build-in adaptability. Future possibilities include a combination of risk management measures including flood defences, resilient development, flood warning systems and emergency responses (POSTnote 342).

Integral to the approach set out in the project is moving away from reactive flood defence, (where flood defences are raised to heights just above the most recent flooding crisis), to proactive adaptive management of future flood risk. This involves a series of timed interventions seeking to manage flood risk within the bounds of acceptability, recognising that if the risks go unmanaged, they will increase in the future, due to the impacts of climate change, along with development pressures on the floodplain becoming more acute, and as existing flood defence assets deteriorate. [261]

However, implementation of appropriate risk management measures at relevant intervals up to 2100 would contain the risk within appropriate levels. This timeline of risk management interventions underpins the concept of a decision pathway,[262] which ensures timely adaptation of the system to manage change levels of risk, dependent on an understanding of a likely future trajectory of the risk and a threshold/limit for future flood risk (such as the current 1 in 1,000 year standard of protection). Different assumptions about the drivers of future change, thresholds or different aspirations for flood risk management, can all generate alternative decision pathways.[263]

These alternative decision pathways can be appraised in terms of their robustness to future uncertainty. The lead time needed to make decisions to adapt to environmental change is often long. For example, it took fifty years from the 1953 flood event for the present-day flood risk management system to be fully operational.[263] These extended timescales highlight the need for monitoring the trajectory of future environmental risks and drivers of environmental change to ensure decisions to adapt are taken within appropriate timescales.

reference point for natural resource managers to judge loss and gains, as once an unacceptable loss of benefits is incurred it is likely to be irreversible.

Risk and Limits

An environmental limit reflects the level of risk that is acceptable to society in relation to loss of ecosystem services. For risk assessment exercises to take better account of where society believes an acceptable level of risk to lie, analytical deliberative approaches can be used to ensure stakeholder participation in formulating acceptable risk levels. The tolerability of risk borne by stakeholders is a critical aspect in ensuring that the communities that bear the risk accept the findings of a risk assessment.

Lay opinion can be used in the framing of the risk problem to be assessed (Box 22) and can be engaged throughout the risk assessment process (Section 4.9). Whether the risk is considered significant is underlain by value judgements that also determine the tolerability of the risk. Ideally, public participation should be engaged early and often in the process. However, deliberative approaches do not guarantee acceptance of risk by communities, although they may improve the legitimacy of decisions and increase the likelihood of acceptance in the longer term.

Whether participatory approaches are feasible depends on the scale at which the decision is being taken. Costs are more acceptable at the community or local decision-making scale but become significant the regional or national scale. The issues that are of environmental concern may also differ significantly at different scales. For example, debates about road building may be locally influenced by effects upon amenity, biodiversity and social issues but nationally they focus more on issues of energy use, climate change and economic development.

By comparison with broader view of how risk assessment is done in other disciplines, regulatory and legislative risk assessment methodology is conservative. However, the Environment Agency has conducted work on engaging communities on the risks arising from coastal erosion and coastal risk management decisions. [264] After the implementation of the Floods and Water Act 2010 through the “National Flood and Coastal Erosion Risk Management Strategy”, local communities will be encouraged to participate in innovative management of flood and coastal erosion risks.[265] Defra also intends to encourage additional local investment by communities, and a greater say for communities in how risks are managed through the strategy in future.[266]

There is a perception that deliberative approaches cost substantially more than standard top down expert-led approaches (the ‘deficit model’ of public engagement[267]), but this may not be true if full costs are taken into account. For example, stakeholder-based decision-making can identify generally-accepted strategies at significant up-front cost, while initially cheaper ‘imposed’ decisions can then prove extremely costly in time and money to defend when contested, as in the case of recent airport and motorway expansions.[268]

Box 22 Public Priorities for Microbial Risk Management[269]

A Defra Research project in the Taw river catchment in North Devon used a “Citizens' Jury” to explore public priorities for managing water quality. A jury of 13 locally-recruited participants sought to gain a better understanding of how risks arising from exposure to pathogenic micro-organisms by way of livestock farming are characterised by interrogating expert witnesses on four key themes:

- the risks arising from the microbial pollution of water courses and their significance (acceptability)
- the origins of these microbial risks and the relative contribution made by livestock farming practices (culpability)
- what more could reasonably be done to mitigate the impact of livestock farming practices on water quality (necessity)
- where responsibilities begin and end when controlling these microbial risks arising from livestock farming (responsibility).

The majority view of the jury was that current risks to human quality of life arising from the microbial pollution of watercourses were relatively insignificant, but it was unanimous in its view that policy makers should take reasonable steps to reduce the probability of microbial water course risks occurring. It also took the majority view that livestock farming played a significant role in contributing to incidents of microbial watercourse pollution compared with other sources. However, in light of the low risks arising, the jury took the unanimous view that prevention measures should centre primarily on the provision of programmes of advice and training. It felt these should be widely disseminated and linked to systems of financial assistance that emphasise low cost and low technology solutions that are in step with existing patterns of farming.

6.2 Assessing the Risks to Natural Resource Systems

Risk assessments are undertaken to inform decisions on how to manage risks. However, as stated in Chapter 5, resilience approaches do not focus on specific risks, but attempt to develop the overall capacity of SES to cope with environmental change. This may require valuing the needs of the whole natural resource system over the vulnerabilities of specific social groups and maintaining the flow of public benefits at the expense of private benefits.

The more systemic view required to create resilience to risks requires understanding of where a natural resource system is vulnerable and hence where the risks are in the system, and how it can be managed to reduce these risks. In summary, their needs to be a greater understanding of:

- the natural resource system's vulnerabilities
- the risk of the natural resource systems vulnerabilities being impacted
- the drivers of the vulnerability in the natural resource system (Section 5.2).

Most risk assessment exercises are locationally and temporally specific, as they relate to developments such as a landfill site or a wind turbine array. Taking a systemic view of risk is more complex and less often done, but is more relevant to environmental limits, in terms of managing systems to be more resilient and less vulnerable to critical thresholds. Public participation could be used in the assessment of systemic environmental risks to natural resource systems. Work on the risk assessment of genetically modified organisms has generated a substantial body of literature about how such exercises should be approached (POSTnote 360).

To maintain natural resource systems within acceptable environmental limits, a robust mechanism is required for assessing whether major proposed projects or policies pose a significant risk of compromising these systems or reducing their resilience to environmental change. However, although it is relatively straightforward to assess risk to specific components of the environment from an impact, such as an individual development project, it is an order of magnitude more complex to assess comprehensively a range of ecological risks arising from environmental change.

Strategies for managing such risks include sharing the (economic) loss of ecosystem service benefits (Chapter 3), mitigating or modifying the drivers of environmental change (Chapter 4), and enhancing the resilience of ecosystems (Chapter 5). If environmental risks are not sufficiently well managed, the only option may be to bear the loss by adapting the way natural resource systems are used and forgoing previous benefits. Once compromised, it may be possible to change the location of the natural resource system being exploited, such as leasing agricultural land or buying fishing quotas in the territory of other countries, but these opportunities can raise their own equity issues.

Ecological Security

Environmental policy is defined not only in terms of concerns for the natural environment versus economic development alone, but also with respect to those aspects of the natural environment that need to be protected to ensure human well-being more generally. Just as there are trade-offs between financial returns and environmental protection, so there are trade-offs between different environmental states and ecosystem service options. For example, there is a choice about whether to allow parts of the coast to be embanked, drained and converted to farmland providing food crop provisioning services or to be restored as intertidal habitats with coastal protection, fisheries and biodiversity benefits.

There is high probability that the 21st century will be characterised by multiple interacting crises, social, economic and environmental, which will impact on human well-being. Current governance approaches tend to contain one immediate crisis and then address the next without tackling underlying chronic problems and risks that create the often-interconnected crises. Without long term management approaches that recognise inherent complexity, there is a significant risk that so called "wicked" issues,[270] such as biodiversity loss and climate change, will impact human well-being.

Ecological security was first promoted as a concept by the US government, but as yet there is no well accepted definition. It can be broadly seen as a condition of ecological safety that ensures access to a sustainable flow of provisioning, regulating, and cultural services needed by communities to meet their basic requirements, such as water and food security and air quality.[271] Ecological security is the inverse function of ecological risk. Increasing the resilience of benefit provision from natural resource systems is a cost-effective means of risk reduction. The consequences of abrupt ecosystem changes on a large scale could affect ecological security to such an extent that it is rational to minimise the risk of breaching ecosystem thresholds, even if there is uncertainty as to exactly where these thresholds lie, given that these changes may be irreversible.

In particular, the long term maintenance of provisioning ecosystem services for food production are critical for human well-being, with ecological security and food security being highly interdependent. A key challenge will be for policymakers to identify and deliver integrated policies that can contribute to food security while protecting and enhancing natural resource systems.

6.3 Food Security

Food security is usually defined as when "...all people, at all times, have physical and economic access to sufficient, safe and nutritious food to meet their dietary needs and food preferences for an active and healthy life".[272] Food security is the outcome of food systems which consist of a chain of activities from production in agricultural or marine ecosystems through processing and retailing to consumption. Each stage throughout the chain can be affected by a range of economic, social and environmental drivers (Figure 19). The interactions among these drivers, activities and outcomes are complex (Figure 20).[273]

Food security has become a heightened policy concern following sharp rises in food prices during 2007/8 in response to multiple factors. These included drought in Australia, world grain reserves at an all time low; a surge in oil prices; subsidies for biofuel production; localised food shortages and reactive policies implemented by governments. Rising oil prices increase fertiliser production costs as well as fuel costs for food production and transport. In addition, where the margin between supply and demand narrows, financial speculation on food commodities may occur, leading to a further increase in food prices.

The continued growth of global population and changes in food consumption patterns pose particular challenges to existing food systems. Food security has been the subject of a recent UK government Foresight study.[274] The three key challenges that need to be met to ensure food security are matching rapidly changing demand for food from a larger and more affluent population to supply, minimising the negative environmental and social impacts of food production and ensuring that the world's poorest people are no longer hungry.[275]

Increasing Food Production

Global food consumption is projected to increase, driven by population growth and changes in diet. The most likely future scenario is that more food will need to be produced from the same amount of (or even less) agricultural land. Of the global 13.4 billion ha land surface, about 3 billion ha is suitable for crop production, about half of which is already cultivated while the remainder supports valued ecosystems such as tropical forest. There is considerable geographical variation in the impact of farming, which depends both on the particular agricultural practices utilised and on the ecosystems affected. While agriculture for food consumption is the predominant land use globally, land is also used for the production of timber, fibre, energy and landscape amenities as well as urban development. Degradation of soil and land through unsuitable use or environmental change is also reducing the area suitable for crop production,[276]

Moreover, there are no new fishing grounds to be exploited with virtually all capture fisheries exploited and most of them overexploited, with global fish catches declining in spite of increasing fishing effort. Climate change and ocean acidification are also likely to affect fisheries resources.[277] However, there is the potential to expand aquaculture production if there are sufficient advances in feed production, resource use efficiency, improved environmental performance and reductions in disease risk.[278]

Sustainable Intensification

Producing more food from the same area of land while reducing the environmental impacts of farming practices has been referred to as 'sustainable intensification'.[279] However, more research is required to assess the effectiveness of different strategies, such as zero or reduced tillage, contour farming, mulches and cover crops. These can improve water and soil conservation but may not improve soil carbon or reduce emissions of the greenhouse gas nitrous oxide. Another example is "precision agriculture", a series of technologies that restrict the application of water, nutrients, and pesticides to only the places and at the times they are required, thereby optimising agricultural inputs. The breeding of crop varieties adapted to lower input farming systems is a further practicable option. All these techniques seek to increase food production by managing key ecological processes and minimising the impact of agriculture on other ecosystem services (section 6.4).

The growth in the human population from about 3 billion in 1960 to 6.8 billion in 2010, coupled with increased income and changes in diet, has been accompanied by substantial increases in crop and animal production. The debate about the most environmentally sustainable means of continuing to increase yield of provisioning services to feed a global population of 9 billion people is likely to continue for decades, with recent studies suggesting that 70 to 100% more food will be required globally by 2050.[280]

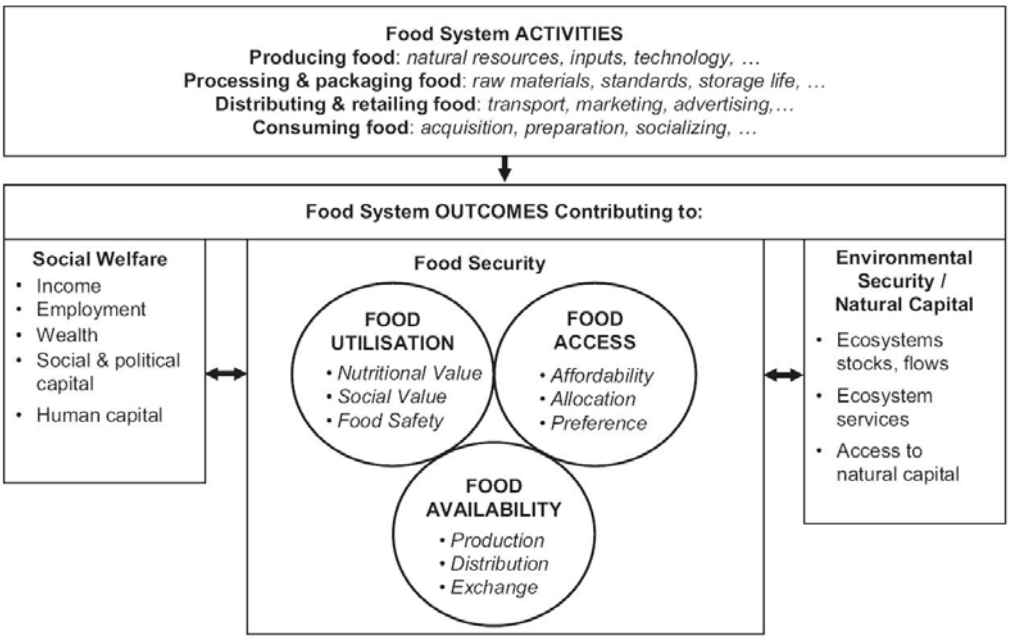
Increasing demand for food will not only be caused by population rise but also by changes in patterns of consumption. Global dietary patterns are being influenced by a complex web of socio-economic trends and drivers. On the demand side, an increasing proportion of the global population lives in cities and often has relatively high disposable incomes. On the supply side, economic growth, regulatory liberalisation, the encouragement of foreign direct investment and globalisation in general has changed food system activities (Figure 18).[281] Regulating use of renewable natural capital to a rate at which its capacity to maintain itself is not exceeded will be challenging.

UK Food Security

The fourth report of session 2008-09 of the House of Commons Environment, Food and Rural Affairs Select Committee, *Securing Food Supplies up to 2050: the challenges faced by the UK*, made a number of recommendations to improve policy frameworks for managing food systems. One was that "the scale and importance of the challenge is such that we recommend that Defra publish a supplement to its Departmental Annual Report, detailing what it is doing to ensure the long-term security of the UK's food supplies, both through trade and domestic production." The committee intended this report to be a first in a series on different aspects of food security. Defra subsequently produced a report on food security in early 2010 in response to the committee's recommendation.[282]

In the UK, 49% of food consumed is currently of UK origin, and the country is self-sufficient in 60% of all food requirements, if exported food is included. It is

Figure 19 How Different Components of Food Systems Contribute to Food Security Including Natural Capital.[273]



almost 100% self sufficient in some food sectors, such as liquid milk, and less than 10% self sufficient in others such as fruit. Improving the environmental performance and increasing food production in the UK is possible, but is likely to result in trade-offs between provisioning services and other ecosystem services provided by landscapes. About 72% of the land area in England is used for agriculture and around 70% of this is used for food and animal feed production, the remainder providing non-food crops such as fibres and biofuels. The amount of land used for non-food crops, such as biomass for energy supply, is projected to increase.

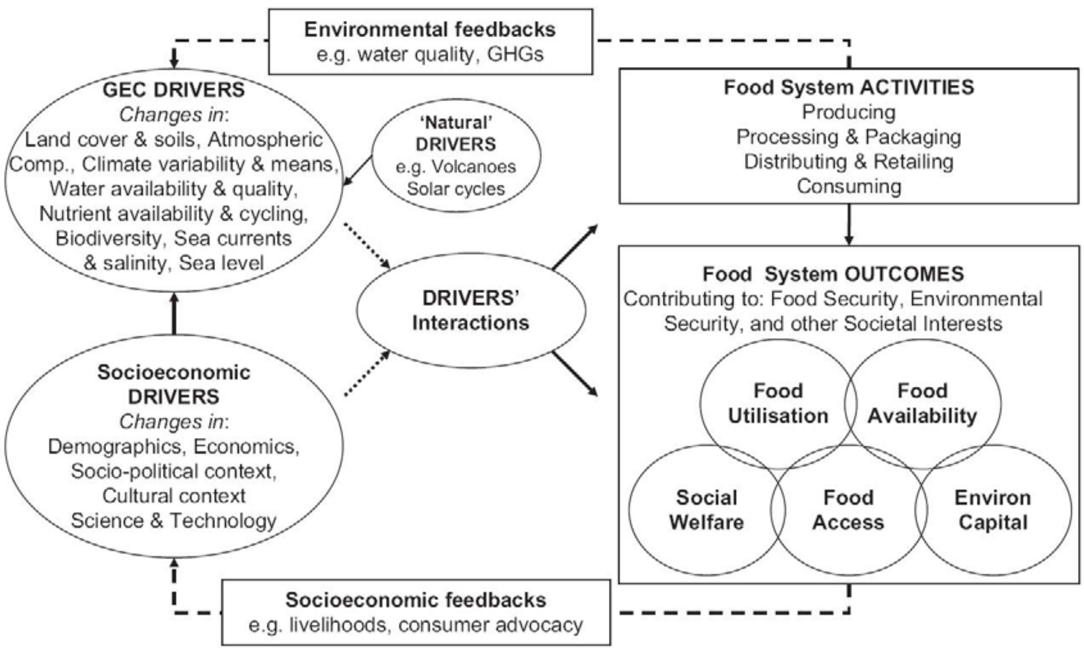
Complexity and Uncertainty in Food Systems
Despite the marked growth in food production in the latter half of the 20th century, there is a substantial risk of reductions in food security in the 21st. Drivers of global environmental and socio-economic change are increasing simultaneously, and involve rapid and complex processes with uncertain outcomes. The local to the global nature of interactions in food systems between processes and actors in different arenas and at different levels, from the local to the regional and international, is introducing greater complexity and uncertainty.[281]

Understanding how to manage the risks to food systems in this context poses considerable research and policy-making challenges. This is further complicated by the fact that food systems are drivers of global environmental change themselves, creating a feedback loop affecting the ability of food systems to be maintained (Figure 19). For example, on many of the low lying coasts of the UK, large areas of intertidal habitat have been converted to farmland by building sea walls and drainage. Given the high cost of maintaining such coastal defences relative to agricultural returns, at many sites it is argued that while this provides a food provisioning service, restoring tidal flooding by the sea would offer benefits in terms of natural coastal defences, sequestration of carbon, sequestration or detoxification of wastes, improved fisheries, higher biodiversity and recreation.

The provision of food security is dependent on many ecological interactions, which may not just be biotic but may be physical, such as measures to increase soil stability. This complexity creates substantial uncertainties and risks and there is the potential for management measures to become dis-benefits that actually increase vulnerabilities, such as introduced biotic control species becoming invasive species (for example, cane toads in Australia). A consequence of modern farming practices is a reduction in the biodiversity of many farming systems with unknown consequences for agricultural ecosystem resilience. If this is sufficiently reduced, there will be an increased risk of abrupt threshold changes with key impacts on levels of provisioning service.

Vulnerability of Food Systems
Food systems can be disrupted and fail to deliver food security as a result of a range of vulnerabilities

Figure 20 A Framework for Food System Analysis Showing the Different Drivers, Interactions and Feedbacks in Food Systems.[273]



resulting from unforeseen abrupt events, inherent structural weaknesses, conflicts between parties in the chain of activities or depletion of natural capital. A key policy concern is how to make the food system resilient to shocks that are unpredictable or uncertain.[281] Vulnerability analysis can be used to determine possible causes of disruption, although they may be the result of complex interactions, with the development decision pathways designed to adapt the system to possible disruption and lessen risks to food security.

Environmental changes can have direct impacts on producing food in a given location. However, the consequences of these impacts for food security are less direct, given that it depends upon many other factors besides availability from local production. The ecosystem services approach can provide a broader perspective on the value of natural capital and on understanding the full societal costs and benefits of different forms of food production, as well as highlighting the risks of natural capital depletion. At a global level, depletion of natural capital will increase vulnerability of food systems, including:

- Water availability at the global level, in the short term. For instance, there are already significant water deficits in key crop-growing regions in South Africa, Australia and the United States. With future demographic changes, there will be increasing competition for water resources between different sectors (urban, industrial and agriculture) and the quality of water resources available for agriculture use is likely to diminish. Recognition of the requirement to maintain freshwater ecosystems, by maintaining environmental flow requirements is likely to reduce further the amount water available for agriculture.[283]
- Other key natural resources in the medium term. The social impact of this may be felt most in farming systems in developing countries. Some such farming methods can result in desertification, reduced soil fertility, and inefficient water use. These lead to diminishing yields for small holder, subsistence or cash crop farmers who do not have access to resources such as fertilisers and pesticides.
- Growing competition for land and energy, in addition to water. This will affect ability to produce food. In the past, the primary solution to food shortages has been to bring more land into agriculture. However, competition for land from other uses such as timber, housing, transport, energy, recreation and tourism, is likely to reduce its availability.[284] Much productive agricultural land has also been lost due to poor land management resulting in desertification, salinisation and soil erosion, a trend that is likely to continue (for example, arable land declined from 0.35 ha/head of population in 1970 to 0.24ha/head in 1994).[285] These factors will need to be offset by reducing waste and increasing the level of yield from each unit of land in production.
- In the longer term, the effects of climate change on the variability of weather patterns may have a substantial impact on food systems, particularly in tropical and sub-tropical regions, although it is not yet possible to provide a robust assessment on the available evidence.[286] Other environmental impacts on agricultural ecosystems, including a reduction in the genetic resource base of crops, will diminish the resilience of these systems to environmental change.

However, the resilience of agricultural ecosystems has not figured to any great extent in the current policy debates on food security. There is an implicit assumption that much production will shift to the northern hemisphere in response to longer-term changes in climate. However, this has not included consideration of how resilient production in these areas will be to the reducing supplies of mineral phosphate and the fossil energy needed to produce nitrogen fertiliser, to the emergence of new diseases, to ozone pollution or invasive species or to rises in the cost of fossil fuels. The available evidence suggests it should be possible to increase food production by 50% by 2050, but this will be reliant on the maintenance of soil fertility and control mechanisms for pests, disease and weeds through advances in agricultural technology.[287]

Food production is fossil fuel dependent, with the manufacture of nitrogen fertiliser the single largest indirect use in agriculture, accounting for more than 50% of total energy use in commercial agriculture. However, while rising fossil fuel prices could pose a major risk to agriculture by causing production costs increase, there is substantial scope for technological and management innovations to increase efficiency and to use alternative energy sources.[288]

Technology could provide some buffering from the effects of environmental change, for example, the use of genetic modification (GM) techniques to deliver the range of genetic adaptations required in plants, such as drought resistance or adaptation to low soil fertility. For example, the Gates Foundation is funding the development of water-efficient maize for Africa in collaboration with a range of organisations.

However, relevant research is constrained by low financial returns and the need to diffuse technology widely rather than to protect patents, and thus would require extensive public funding. This would be in addition to the growth in agricultural research funding required to increase yields at a rate sufficient to meet increasing demand.[289] In addition, there is some public acceptance for the use of GM technologies in some developed and developing countries.

Agricultural Ecosystems

Agricultural ecosystems are natural capital assets, and the flow of services that they provide is the 'interest' on that capital. As agricultural ecosystems cover nearly 40% of the terrestrial surface of the planet,[290] there is an increasing requirement for ecosystems to provide not only food provisioning services, but also a range of other ecosystem services, that are either of direct benefit to agriculture (such as pollination) or which provide other benefits to society (such as water quality). In the EU, food is primarily produced in intensively managed agricultural ecosystems which cover 45% of the its land area and have a total annual economic value of around €150 billion.[291]

Just as investors choose a portfolio of produced capital to maximise the return on that capital over a range of market risks, so managers and policymakers need to choose the mix of genes, species, communities, and ecosystems that will enhance the flow of ecosystem services from agricultural ecosystems over a range of environmental and social risks.[273] This not only includes provisioning services, but other benefits such as flood control, water quality, carbon storage, disease regulation, waste treatment and cultural services such as iconic landscapes. Management actions can also result it in potential disbenefits, such as nitrification and sedimentation of watercourses, greenhouse gas emissions and loss of biodiversity (Figure 20).

This requires understanding the risk implications of changes in that mix to design adequate strategies for management and conservation to maintain resilience in agricultural ecosystems. Most of Europe's remaining biodiversity exists within a mosaic of heavily managed land and highly exploited seascapes, created by agricultural, forestry and fishery practices. Many other policies in Europe that are not biodiversity policies nevertheless have an important impact and may contribute to conserving, managing and restoring biodiversity. For instance, the CAP and WFD are relevant to the status of agricultural ecosystems (Chapters 4 and 5).

Almost all landscapes in the UK have been modified by agricultural activities and most natural, unmanaged ecosystems sit in a matrix of agricultural land uses. To maintain the resilience of agro-ecosystems, there is a need to conserve existing biodiversity and ecological processes in agricultural landscapes and to adopt practices to increase the environmental sustainability of agricultural production while ensuring viable economic gain (Figure 20).

For example, pollination services depend on the movement of organisms across agricultural landscapes, with pollinators moving among natural and semi-natural habitats in diverse landscapes that provide them with necessary habitat and resources throughout their lifecycle (Section 5.5). Landscape structures simplified by agricultural intensification, resulting in the loss of field margin vegetation and remaining natural habita,t may diminish levels of this ecosystem service.[292]

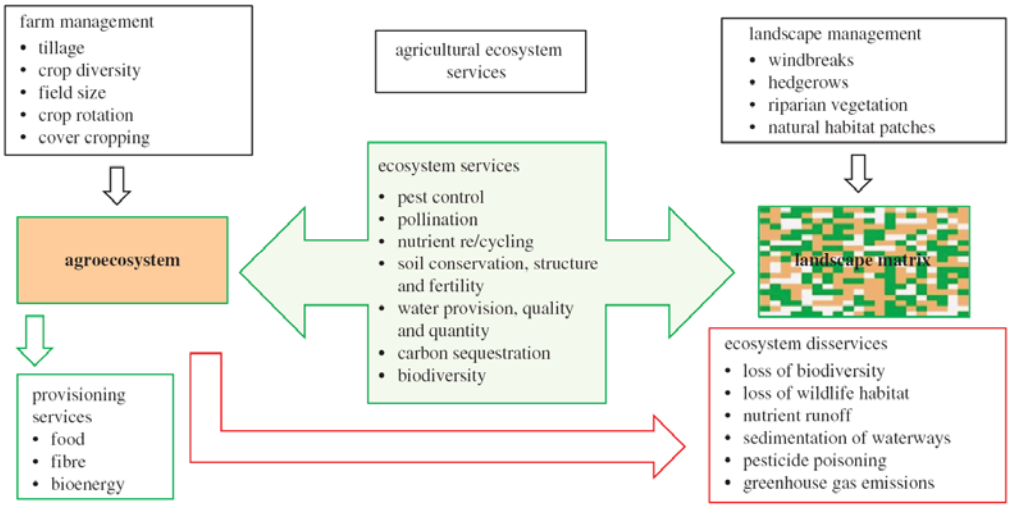
The agro-biodiversity research strand of the international Diversitas multidisciplinary biodiversity research programme is aiming to establish the scientific basis needed to address the trade-offs between food production, biodiversity conservation, ecosystem services and their implications for human well being in agricultural landscapes. The emergence of new crop diseases and weed species highlights the importance of maintaining the genetic diversity of crop species including that of landraces (local or traditional varieties of domesticated animal or plant species and crop wild relatives). However, as regards the levels of other ecosystem services that need to be maintained to support production, far less is known.[293]

Unexploited genetic material from wild relatives of plant crop species and from rare livestock breeds or wild relatives are important resources for allowing genetic adaptations to environmental change to be used. International collections and gene banks are respositories of such genetic variation, but further actions are required to ensure that locally-adapted crop species and livestock are not lost, including efforts to maintain these resources in the semi natural or farming systems in which they occur and to which they are adapted.

Ecosystem Service Declines

In the longer term, declines in provisioning services will be driven by declines in regulating and supporting services, in addition to unintended impacts of technology. For example, pollination services (Box 2) will be a key food security issue, along with other regulating ecosystem services

Figure 21 Impacts of Farm Management and Landscape Management on the Flow of Ecosystem Services and Disservices To and From Agroecosystems.[294]



such as soil quality (Box 6), pest regulation (Box 4) and water availability. Perennial vegetation in natural ecosystems such as forests can regulate the capture, infiltration, retention and flow of water across the landscape. The plant community plays a role in regulating water flow by retaining soil, modifying soil structure and producing leaf litter. Forest soils tend to have a higher infiltration rate than other soils. Forests help to reduce peak flows and floods while maintaining the level of flows in watercourses (Boxes 14-16) and have low soil erosion rates, maintaining water quality.[295] Water availability in agricultural ecosystems depends not only on infiltration and flow but also on soil moisture retention, regulated by plant cover, soil organic matter content and the organisms in soils (Box 6). Management measures, such as the retention of woodland around water courses, good soil management and farm storage of rainwater, can significantly reduce the impacts of agriculture on water quantity and quality.[296]

Although farmers benefit from regulating and supporting services, many ecosystem services are public goods. The management of agricultural ecosystems can influence the delivery of services to individuals who do not control the production of these services, such as the quantity and quality of downstream water supply. The costs of the loss of services, such as water quality, also accrue to the users rather than the land managers. Information on the factors that influence the quantity and value of ecosystem service benefits is critical for optimal land management policies and to create incentives that are adaptable to environmental change and changes in market demands for benefits.

Farmers have a direct interest in managing ecosystem service benefits that are provided at the farm or field scale, such as soil fertility or pest control. Benefits provided at a larger scale on a shared basis, or that accrue to others, such as those that require complex landscapes with areas of natural and semi-natural habitat, require additional incentives. Studies suggest that agricultural ecosystems can support many ecosystem services while still maintaining or enhancing food provisioning services.[296]

Natural England stated in its policy on food security and the environment that the CAP should aim to provide environmental security through the management of the ecosystem goods and services underpinning soil, air, water quality, biodiversity and cultural landscapes. It also stated that it should encourage farming practices that contribute to long term food security and other ecosystem service benefits.[297] It is likely that enhanced understanding will lead to improved ecosystem service benefit provision in agricultural landscapes in Europe, but growing demand for food, environmental change and displacement of agricultural ecosystems by urban development are likely to lead to increasing ecosystem disbenefits in developing countries.[298]

Resilient Food Systems

Coupled social and ecological systems, like food systems, highlight that slow variables, such as changes in regulatory services, are critical to ensuring a system's resilience. Such approaches also reveal the key role governance and other institutions play in negotiating the use of social and ecological resources. Understanding the coupled nature of food systems can help managers and policymakers to choose options that are likely to increase rather than decrease the resilience of the system.

Expanded trade and a diversity of supply chains can provide insurance against regional shocks in agricultural production such as conflict, epidemics, droughts or floods, shocks that may be increased by drivers of global environmental change. However, according to systems theory, a more highly connected food system may be more prone to instability such as the propagation of economic perturbations. In contrast, the balance of evidence is that natural ecological systems have increased resilience when the biotic components are highly connected.

Increasing the resilience of food systems will have trade-offs in terms of economic efficiency and social justice. For example, approximately one-third of global cereal consumption is fed to animals, reducing its availability for other uses. Ethical and efficiency concerns have led to calls for governments to mitigate increasing demand for meat and dairy products through policy measures to encourage the most efficient and least environmentally-damaging livestock systems. Although meat and dairy consumption has stagnated in developed countries, it is likely to continue to increase in developing countries.[299]

As well as the social and political constraints on instituting such policies, it should also be noted that a substantial proportion of livestock is grass-fed, and some is fed on waste. Livestock waste is also a key fertiliser in many agricultural systems. The complexity of the food system means that any measures to increase resilience will need to be situation- and scale-specific.

Measures to increase the environmental sustainability of agricultural ecosystems do not necessarily reduce yields or profits. One study of 286 agricultural sustainability projects in developing countries, involving 12.6 million mainly smallholder farmers on 37 million hectares, found an average yield increase of 79% across a very wide variety of systems and crop types.[300] Understanding the optimal regulation of food systems emphasises the need for taking a “competing risks” approach in trading off costs and benefits.[301]

Drivers, such as climate change and population growth will impact the food system and need to be managed so that they do not have unintended impacts on other factors within the system. Food production also needs to be treated within the broader context of one of several competing uses for natural capital, such as freshwater, land and energy. As with all SES, greater understanding of food systems will also arise not from not a single academic discipline, but will require combined approaches, including natural sciences, economics and social sciences.

There are also social trade-offs with the excessive consumption of energy-dense/nutrient-poor foods, with more overweight obese people globally than underweight or malnourished people, a change which has occurred in tandem with the globalisation of the food system.[302] The consequent health burden arising from this change is a trade-off against the increased resilience of a globalised food system. A key challenge is balancing these different outcomes to ensure optimal outcomes while minimising unintended impacts. This can be achieved only by consideration of the system, rather than its individual components in isolation.

Monitoring Food Systems

A critical area for negotiating the necessary trade-offs between increased food production and reductions in other ecosystem service benefits is the definition of relevant indicator sets to allow alternative strategies and decision pathways to be assessed. Indicators to show the resilience of agricultural ecosystems are being investigated but the relevant metrics are still under development.

To do this for environmentally sustainable food systems, encompassing all relevant activities will be considerably more complex. For example, complex food products may contain ingredients sourced from around the globe that have undergone many different processes. The Global Environmental Change and Food Systems (GECAFS) programme has funded research to develop frameworks for assessing food systems and determining the likely impacts of policy choices on these systems, piloting case studies in areas such as India, the Caribbean and South Africa.[303]

GECAFS is also researching methods for determining the optimal adaptation to environmental change choices for food systems. The example shown in Figure 21 illustrates how semi-quantitative evidence of the impact of afforestation policies on agricultural land would impact food systems and other key ecosystem services. However, there is a risk that over-emphasis of relatively simple-to-measure environmental impacts, such as carbon emissions or water use, may lead to other aspects that are harder to quantify, such as impacts on regulating ecosystem services, not being considered.

Negotiating Trade-offs
If appropriate metrics can be derived, one means of negotiating these trade-offs would be through directly informing consumer choice. At present, consumer surveys in developed countries suggest that price, convenience and safety are central to food choice, but other values have also become significant, such as ethical choices about how food is produced and concerns about food security.

NGOs, including the WWF, and government advisory bodies have argued for a pooling of separate food system criteria to allow consumers to make informed trade-offs in their consumption decisions. The UK's former Sustainable Development Commission suggested the creation of "omni-standards" to inform consumers about what constitutes a healthy sustainable diet.[304] Suggested criteria for these are shown in Figure 22.

The diverse relationship between, and roles for, governance institutions, industry and civil society make the negotiation of trade-offs in food systems challenging, as does the range of concerns each party must trade off. However, the pressures placed on food systems by global environmental change and the implications for food security if environmental limits are exceeded in natural resource systems critical to production, will increase the need to make trade-offs explicit to consumers and regulators.

6.4 Managing Environmental Limits

The impacts of human activities on natural resource systems and related ecosystem services are reaching levels at which ecosystems maybe changed in ways that significantly impact human well-being (Chapter 2). Current approaches to issues such as land use, urban infrastructure and water management are being increasingly questioned in the light of interconnected challenges faced in the 21st century.[305] Avoiding the breaching of environmental limits is becoming a key challenge for policy-makers at all levels of decision making (Chapters 3 and 4).

Environmental Policy Integration

Governments have taken steps to increase policy integration and coherence in areas such as EU fisheries and agriculture policies and there are initiatives to have greater consideration of biodiversity and ecosystems in other policy areas such as planning, transport, finance and trade. Environmental Policy Integration (EPI), which seeks to address the drivers of environmental change by integrating environmental considerations into design and adoption of policies in all sectors, is a key aspect of sustainable development policy frameworks. Environmental assessments of programmes, plans and projects, such as EIA and SEA (Annex D) are also critical tools at the project and plan level of governance.

The UK has had various EPI mechanisms since the 1990 white paper on the environment. The previous government established two independent bodies to examine EPI and sustainable development policy in the UK, the parliamentary Environmental Audit Committee in the House of Commons (EAC) and the Sustainable Development Commission (SDC). The present government has withdrawn funding for the SDC and intends to establish alternative mechanisms within government departments. The concerns raised by the EAC concerning EPI integration into Regulatory Impact Assessments (RIAs) led to regular annual scrutiny by the National Audit Office since 2006 for coverage of environmental sustainability issues.

Figure 22 Spidergram Showing the Likely Proportionate Increases and Decreases in Ecosystem Service Benefits and Food Production and Consumption in Response to Afforestation Policies.[306]

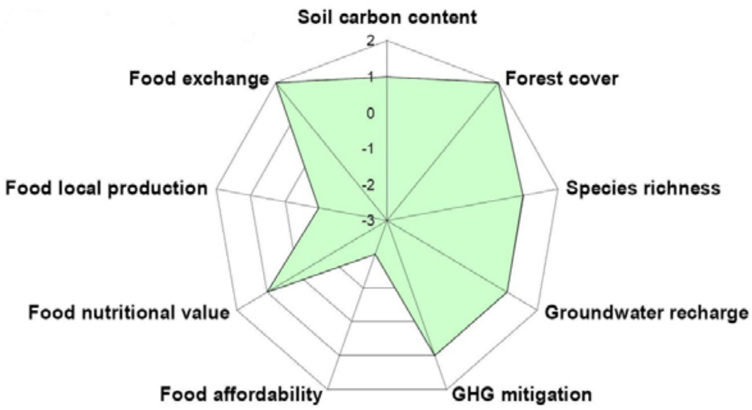


Figure 23: Possible Criteria for Omni-standards Split Into Four Categories

'Omni-Standards' or 'poly-values',
[which all have to be met not traded off]

Quality:
- Taste
- Seasonality
- Localness (?)
- Fresh (?)
- Identity / authenticity

Environmental:
- Climate change
- Water
- Land use
- Biodiversity
- Waste reduction

Social values:
- Pleasure
- Animal welfare
- Working conditions
- Equality
- Cost internalisation
- Trust

Health:
- Safety
- Nutrition
- Access / affordability
- Information & education

These categories would be complex for consumers to negotiate, so may need to be simplified for labelling purposes

The current system of EPI is integrated into Impact Assessment processes, (which replaced RIAs), mandatory for all significant legislation, overseen by the Department for Business, Innovation and Skills. Impact Assessment assesses the economic costs and benefits of new regulation, aiming both to avoid unnecessary or overtly costly regulation and to improve the design of policy. This is intended to decrease the costs of regulation for both business and public authorities, thereby enhancing economic competitiveness and contributing to stabilisation of public spending.

Defra is responsible for the specific environmental impacts test and sustainable development impact tests within the Impact Assessment, which include consideration of impacts on ecosystems (Chapter 4, Annex D). The EAC and NAO will scrutinise departmental impact assessment approaches on parliament's behalf to determine the coverage of environmental impacts. Legislators can play a role in considering trade-offs between economic growth and the natural environment, but require governments to supply appropriate information, such as ecosystem service valuations.

However, it should be noted that it is difficult to determine what has flowed from previous EPI activity in terms of concrete policies that address the drivers of environmental change.[307] The impetus for broader and more integrated forms of policy appraisal is supported by the promoters of environmental policy to facilitate sustainable development. Critics have noted that it might actually form the basis for other government departments to challenge environmental policy on competitiveness grounds, if the appraisal process is dominated by the deregulation agenda.[308]

Responding to Complexity

It is not the intention of this report to imply that sufficiently enlightened technocrats supplied with the appropriate evidence base and the relevant toolkit can provide win-win policy solutions through improved policy integration, transparency and accountability. There is an emerging literature, for instance, on the difficulties of using instruments such as policy appraisal systems to achieve sustainable development objectives and the institutional constraints within government that limit the influence of such approaches.[309]

It can be argued that increased use of evidence of impacts of ecosystem services in policy and decision making, as set out in Chapters 3 and 4, may be equally ineffective. Nor is it clear whether adjusting the machinery of government is more effective at reducing the risk of exceeding environmental limits and maintaining ecological security than, for instance, regulation to tighten environmental standards or targets.

The complexity of interactions between humans and the environment implies that all-encompassing solutions may not exist or may not be politically achievable, given the constraints. Optimal policy outcomes from an environmental limits perspective are those that increase the resilience of natural systems to environmental change, but this will not always be compatible with economic and social objectives.

Even minimising the extent to which the risk of exceeding environmental limits increases will involve politically difficult compromises. The decision support tools for ecosystem-based management could be used within adversarial policy frameworks to shut down political debate and apply environmental limits set out in national or European legislation. Equally, they could be used to map and inform areas of political debate to build consensus about how best to balance levels of natural resource use and protection, as is the intention of the Ecosystem Approach.

This would require the development of flexible institutions of governance and adaptive co-management frameworks that facilitate local organisations interactions with each other and with other organisations at different levels (Section 5.4). Although consensus-building and public participation can be both expensive and time consuming, they are complementary to environmental citizenship approaches. These engender pro-environmental behaviour in public and in private, driven by belief in fairness in the distribution of environmental goods, in participation and in the co-creation of ‘sustainable’ policies.[310]

Local and Global Frameworks

The most appropriate governance level for ecosystem based management is at the scale of the ecosystem, such as the river catchment level. At these scales, participation of users and beneficiaries of natural resource systems in governance is usually feasible. The defining of environmental limits requires an understanding of the aspects of natural resource systems critical to human physical, psychological and economic well-being at an individual, community and societal level. These may be most straightforwardly defined at a local level, but they also need to be identified at national and global levels, to reflect collective human impacts on the environment and the consequences for collective human well-being, such as reductions in food security.

It is not clear whether international governance mechanisms can provide the basis for agreeing and enforcing global environmental limits, given the failure to secure a climate change framework to replace the Kyoto Protocol, adopted at the third session of the Conference of Parties to the UNFCCC (COP 3) in 1997 in Kyoto, Japan. Opened for signature at the Earth Summit in Rio de Janeiro in 1992, and entering into force in December 1993, the Convention on Biological Diversity (CBD) is an international treaty for the conservation of biodiversity, the sustainable use of its components and the equitable sharing of the benefits derived from the use of genetic resources.

In April 2002, the parties to the convention committed themselves to achieve, by 2010, a significant reduction of the current rate of biodiversity loss at the global, regional and national level, but it continues to increase. Although more focussed, it is not clear whether the “Aichi target” agreed at the last CBD meeting in October 2010, to decrease the rate of habitat loss by 50% by 2020, while increasing the land area to be protected from 13 to 17% and marine protected areas from 1% to 10% over the same period, will be any more successful. The most appropriate policy frameworks for slowing global rates of environmental degradation will be the subject of further international negotiation at the Rio +20 earth summit in 2012.

Adapting to Environmental Change

A critical aspect of ensuring ecological security is acceptance that future global environmental changes are an inevitable consequence of the impacts of past human activities, such as existing levels of greenhouse gases or the transport of invasive species, and that these changes will often have uncertain consequences. Natural resource systems could, however, be managed to ensure resilience and recovery from these consequences, if sufficient understanding of these systems is achieved (Chapter 5).

A scientific evidence base that can inform the concept of ecosystem service decision making (Chapter 4) is being developed, but is still in its early stages. An integrated perspective across ecosystems and the human drivers of changes within those ecosystems is required. However, it is not always straightforward to determine the interrelationships between different factors and the complexities involved in measuring ecosystem services at scales relevant for decision making, both geographical and temporal, are significant (Annex B).

The vulnerabilities in coupled SES, such as food systems, need to be further researched on a systems basis before the risks of exceeding environmental limits can be quantified. With improved information on specific risks, natural resource systems and related ecosystem services could be managed through a series of interventions to maintain risk within tolerable levels, by measures either to maintain resilience or to reduce the drivers of change on the system.

How the impacts of drivers of environmental changes at the global level are manifested at the UK level is likely to remain uncertain. A systems approach, combined with adaptive management, provides a framework for understanding the dynamics of coupled SES at a range of scales and for ensuring the resilience of the system to unpredictable challenges.

Annex A Pressures on Natural Capital

A1 Biodiversity Loss

Globally, none of the world's areas with high levels of biodiversity ('biodiversity hotspots') have more than one third of their pristine habitat remaining.[311] UK biodiversity has declined substantially since the advent of the industrial revolution, with the alteration of natural habitats to meet growing demand for food, energy and infrastructure. As summarised in Chapter 2, most of the UK's biodiversity exists within a mosaic of heavily managed land and exploited seascapes. The government is seeking to increase policy integration for biodiversity across a range of sectors (such as trade, planning, transport and finance) through the Impact Assessment process (Annex D) and the wider application of the Ecosystem Approach (Chapter 4). However, there are significant knowledge gaps in the links between biodiversity loss, ecosystems services and human well-being.[312]

Functional Traits and Ecosystem Services

A "functional trait'" is a characteristic of an organism, such as being a predator, which is linked to the function it performs in a community or ecosystem. In other words, it describes how it is connected to other organisms or to the wider environment, in terms of matter and energy flow, or how its behaviour is influenced or influences other organisms or abiotic components of the ecosystem.[313] These "functional traits" can include population sizes of organisms, the role of a particular life stage of species, their spatial distribution, a range of behavourial and physical traits relevant to ecological processes and the diversity of species that perform similar roles.[314]

For example, predatory insects can perform important pest control functions, such as the predation of aphids by hoverflies, lacewings and ladybirds. The degree of feeding specialisation, and the rates at which they consume their prey species and at which they reproduce, determine how effectively they can act as a biocontrol. It is estimated that insect predators and parasitoids account for 33% of natural pest control for some crops, the economic value of these services equating to approximately $4.5 billion globally.[315]

Although functional traits and their role in ecosystems have mainly been studied in plants, there is evidence for the importance of the interaction of traits across different parts of the food chain in the control of many ecosystem services.[68] Functional traits can be split into two groupings, "effect" traits, which affect ecosystem function, such as being a predator, and "response" traits, which affect the response of organisms to environmental factors.[316] Existing evidence suggests that if there is an overlap or linkage between the 'response' and 'effect' traits, there is a greater likelihood that environmental change will impact levels of ecosystem service provision. [317]

Many ecosystem services depend on several ecosystem functions carried out by multiple parts of a food chain, for example carbon is sequestered in soil as a complex outcome of several ecological processes including wood production, litter decomposition and microbial carbon immobilisation, each dependent on a series of functional traits of plants and interactions with soil biota.[318]

Although ecosystem services are generally understood to be properties of whole ecosystems, ecological functions that support services often depend on particular populations, species, guilds of species (Section 5.3) or habitat types, with the loss of a functional group resulting in the loss of an ecosystem service.[319] Identifying and quantifying the organisms necessary for service provision (the 'Service Provision Unit', SPUs) and the other aspects of biodiversity that support them are critical to maintaining services and could inform relevant planning and conservation policy decisions. A recent review of the quantitative evidence showed links between the traits of organisms and the provision of ecosystem services in 500 examples.[320]

The identification of functional traits in different parts of food chains and quantifying their contribution will be critical to understanding the role of biodiversity in ecosystem service provision, as well as to identifying vulnerabilities arising from these interactions. For example, climate change could simultaneously affect plants and pollinators, with impacts on ecosystem service benefits such as food production.[321] Key functional traits include climate tolerance, growth rate and efficiency of nutrient use.

Functional Diversity

The notion of functional diversity refers to the number of functional traits or groups that might be present in a particular ecosystem.[313] The amount of functional diversity required to maintain processes differs between ecosystems, but where there is a multiplicity of species present performing similar functions, they are believed to act as a buffer to physical and biological changes in the environment. Some species, including ones which are currently rare, may have functional properties which mean that under changed circumstances they take on a prominent role in the ecosystem and therefore ensure its long term resilience.[313]

There is already evidence that functional diversity may decline more rapidly than species diversity in response to land use change.[322] Ecosystem management and exploitation can select against certain traits. For example, the use of fertilisers filters out plants capable of growing on low fertility soils. Plants with generalist traits that dominate disturbed habitats will be favoured at the expense of species that grow in more stable habitats.

There is also growing evidence for the importance of these interactions as causes of complex responses of ecological processes to environmental change.[323] Insufficient diversity of organisms to buffer environmental changes increases the risk of "ecological surprises". These can involve disproportionate, large, unexpected, irreversible, and negative alterations of ecological processes, which may affect ecosystem service benefit provision. While it is difficult to generalise about how particular ecosystems will respond to changes in the abundance of species or groups with particular traits or characteristics, some conclusions can be drawn, including:[313]

- There is evidence that particular combinations of species can interact in a complementary fashion to increase average rates of productivity and nutrient retention.
- The vulnerability of communities to invasion by alien species is influenced by species composition and under similar environmental conditions, generally increases as species diversity decreases.
- Ecosystems subject to disturbance can be stabilised if they contain species with traits that enable them to respond differently to changes in environmental conditions.

Direct and Indirect Interactions

The alteration of direct interactions between organisms such as predation or competition, by human impacts, is well understood. For example, sea otters prey on various marine invertebrate organisms including sea urchins, which graze on the lower stems of kelp seaweed, causing the kelp to drift away and die. Sea otters were hunted to near extinction off the north-east coast of America, which in turn led to the degradation of coastal kelp-based marine ecosystems in open waters due to the overgrazing of kelp by sea urchins. Effects included the extinction of other species found in kelp habitats and increased coastal erosion due to increases in wave velocity reaching the shoreline. It is relatively straightforward to deduce from such direct interactions when organisms are playing a key role in functions that underpin ecosystem service provision.

The links between some ecosystem services and species are much less well understood as they may arise from a large number of direct and indirect interactions between species and with their physical environment. Functional traits of organisms also interact for mutual benefit less directly, for example, a plant can benefit neighbouring plants by providing shade and nutrients, and protection from extremes of temperatures or from impacts of browsing animals. In particular, there appears to be a strong link between basic ecosystem properties such as species richness and functional diversity and supporting ecosystem services, such as primary production.[324]

Long term experiments on grassland species indicate that species richness increases productivity as result of such indirect beneficial interactions.[325] The performance of a plant species can either be enhanced by the presence or action of another species (facilitation) or plant species can minimise competition with each other through sequential timings of their flowering and vegetative phases (complementary interaction).

In general, such indirect interactions are less straightforward to define, measure and monitor, but may play an important role in buffering the impacts of environmental change. A better understanding of direct and indirect interactions within and between parts of food chains will give insights into how these interactions may buffer or amplify the effects of environmental change on ecosystem service provision and the likelihood of ecological thresholds being exceeded.[323]

Maintenance of Ecosystem Service Provision

A diversity of organisms with similar traits is a better predictor of the maintenance of ecosystem service outputs than diversity in species alone. To maintain ecosystem services, a wide range of species with relevant traits may need to be conserved.[326] An increased diversity of species, and hence direct and indirect interactions, appears to make ecosystems better able to withstand environmental changes, such as those in levels of nutrients and invading species, due to the differing (asynchronous) species' ability to adapt to the change applied.[327]

The available experimental evidence on grasslands suggests that about ten plant species per square metre are required to maintain ecological processes and structures in temperate regions with a much higher overall number of plant species required at the landscape level.[328] However, such approximations of biodiversity requirements are underestimates, because they are based on limited understanding of the ecological functions and processes involved.

The relationship between numbers of species and measures of ecosystem processes such as nutrient cycling remains unclear. Evidence from grassland experiments suggests that greater numbers of species are required the more the ecosystem processes that are supported by an individual ecosystem,[329] because while some species may contribute to primary productivity, a different set of species may play a key role in nutrient cycling. However, there is a lack of understanding about how sensitive ecosystem service delivery is to changes in processes caused by change in species numbers and identity.[330] Identifying relationships between functional traits and ecosystem services could provide a basis for managing biodiversity for the provision of multiple ecosystem services.[331]

Adaptation to Environmental Change

Natural systems are not infinitely flexible with respect to environmental changes, and the ability of ecosystems to respond to change will depend on their soils, water and other physical components, as well as the extent to which the physiology and ecology of the species present permit them to cope. For example, the vulnerability of landscapes to climate change can be estimated through the degree of climate change exposure, the sensitivity of the semi-natural habitats present and the biological and physical capacity of organisms and ecological processes to adapt.[332]

Vulnerability approaches (section 5.2), used in global change biology and climate impact research biology, assume that some systems will be more sensitive to change than others and that there are varying degrees of resilience to change inherent within different systems.[333] Those habitats most sensitive to climate change impacts, such as temperature increase and drought, include:[334]

- lowland meadow
- upland heath
- lowland heath
- fens
- lowland raised bog
- blanket bog
- purple moor grass and rush pastures
- coastal and floodplain grazing marsh
- lowland beech and yew woodland
- wet woodland.

The recent “Lawton Review” recommended that responsible bodies should revise conservation objectives for SSSIs and other wildlife sites to respond to the effects of climate change, in particular by aiming to enhance habitat diversity and to support underpinning ecological processes, while taking account of the requirements of current species and habitats.[57] The adaptive capacity of landscapes in the UK to climate change has been assessed using biological and physical measures of landscape heterogeneity. Measures include number of soil types and diversity of land cover types, and permeability to species movement, allowing the areas of landscape to be mapped and classified for vulnerability. Heterogenous landscapes are more likely to contain a greater range of microclimates and environmental conditions allowing species to persist.[332]

Biological Limits to Adaptation

Many species’ populations exist not as spatially isolated groups, but as metapopulations, (sets of local populations linked by the dispersal and movement of individuals to adjacent populations).[335] Examples include bumblebees, butterflies, amphibians and freshwater molluscs.[336] In such metapopulations, if the linkages between habitats are lost, due, for example, to land use change, surviving populations on adjacent habitat patches often decline, even if the remaining patches retain their good condition.[337]

A key consideration is the diversity of functional response traits that different species have to a particular environmental pressure (response diversity). This varying ability to adapt to environmental change is critical to whether and how long ecosystems take to recover from disturbance. The functional “response” traits of species present in habitats determines their reaction to environmental changes and, in turn, whether ecosystem service provision is affected. The ability of populations of organisms to cope can be achieved through changes in the genetic composition of a population (evolution), the capacity of individual organisms to vary their form and structure, behaviour or metabolism (plasticity), or abilities to move to other habitat (dispersal).

The ability of an organism to respond will vary according the rate, extent and nature of environmental change, and this ability will vary among species, habitats and geographical areas. If none of these strategies is available or effective, the population of species that perform functions within ecosystems will die out. If that function is not performed by other species that are able to adapt, that function and its associated ecological processes will be altered and ecosystem service provision will be affected. The concept of resilience to external pressures is further discussed in Chapter 5.

There are insufficient research data at present to determine the biological limits to adaptation to environmental change and the likely fate of natural populations under different environmental conditions.[338] It is rare to find a linear causal path from changes in drivers of biodiversity loss, through changes in biodiversity and ecosystem processes, to impacts on ecosystem services and human wellbeing, due to the complexity of the issues involved.[339]

However, present rates of extinction imply that there is a significant risk of these thresholds being exceeded for many populations of organisms (Section 2.5). Although at an aggregate level the current rate of biodiversity loss suggests that thresholds are being approached, it would be extremely difficult to define a threshold of species loss. At an aggregate national or regional level, current levels of understanding are not sufficient to deduce the likely effects on ecosystem service benefit delivery and to define environmental limits for species loss accordingly.

A2 Land Use Change

Vegetation is the primary interface between humans and the biosphere through land cover conversion (changes to the physical material at the surface of the earth such as trees or water). The land use change transforming the terrestrial surface of the earth is driven by a diverse array of activities including:

- clear-cutting or burning of original forest cover;
- agricultural activities such as ploughing, application of pesticides, manufactured nitrogen and phosphorus fertilisers, cultivation of favourable plant species and grazing of domesticated animals;
- creation of more homogenous landscapes through conversion to human use;
- transfer of organisms between locations; and,
- the construction of buildings and infrastructure.

Land use is the principal driver of environmental change and has become the major factor in the distribution and functioning of ecosystems and the provision of ecosystem services. Land is a finite natural resource and there are competing demands upon it in terms of uses and for the services it provides. Land cover conversion from natural vegetation communities to human uses has a range of impacts on the environment at different spatial scales. It can fragment habitats and change types of vegetation cover, ecological productivity and the amount of biomass produced and impact on biogeochemical processes for water, carbon and nitrogen.[340] In particular, the conversion of vast extents of primary forest in the subtropics and tropics strongly reduces the capacity of these ecosystems to regulate regional climate, long-term soil fertility, and water quality, quantity and runoff (Box 21).

A better understanding of how land resources contribute to wellbeing and whether changes in land uses are desirable is critical to discerning where the acceptable limits of land resources lie. The main land-cover types in Europe are forest, 35%; arable, 25%; pasture, 17%; semi-natural vegetation, 8%; water bodies, 3%; wetlands, 2%; and built infrastructure, 4%. The general trend is for built infrastructure to expand at the expense of land cover types, by 0.1% in the period 2000 to 2006.[341] The UK National Ecosystem Assessment published in March 2011 will provide a comprehensive assessment of the impact of different land uses on ecosystem service provision in the UK.

Along with providing food, fibre and fuels, agricultural ecosystems supply other vital services, such as pollination and natural pest control. Less diverse crops, simplified cropping methods, use of fertilisers and pesticides and homogenisation of landscapes all have negative effects on biodiversity. Seventy percent of species of "European interest" are linked to agricultural ecosystems, with agriculture the main land use in the EU-27 (47% of the territory). Several pressures from agriculture have been addressed by reducing nitrogen inputs and through the implementation of agri-environment schemes (AES).[342]

Soils

Soil supports agriculture, wildlife and the built environment, filters water, stores carbon, and preserves records of the ecological and cultural past. Soil degradation involves both physical loss (erosion) and the reduction in quality of topsoil associated with nutrient decline and contamination. It can be the result of one or more factors including:

- physical degradation (erosion, loss of structure, surface sealing and compaction);
- chemical (pollution) and biological (loss of soil organic matter and biodiversity) degradation; and,
- climate and land use change (which may accelerate the above factors).

Box 23. Deforestation

Deforestation is technically defined as "the reduction of tree canopy to less than 10% crown cover".[343] Most tropical deforestation results from clearing of space for agricultural land. This reflects the fact that it is normally more profitable to clear forest and grow crops, than it is to harvest timber and other forest products sustainably.[344] Currently, tropical deforestation is largely caused by demand for subsistence food crops, especially in Africa, but in Latin America commercial cattle ranching and soya cultivation are significant drivers. In SE Asia, palm oil and wood pulp production, along with large scale timber extraction are also important. However, underlying these direct causes of deforestation are issues of economic development, land ownership and governance, which have stymied previous international efforts to reduce deforestation. Reliable data on the causes of deforestation do not exist, partly because of monitoring problems, but also because different causes of deforestation are often entwined. The estimates in Table 1 are known to be highly uncertain. A typical sequence of deforestation in a Latin American rainforest might start with new access due to a road being built, followed by selective logging of the valuable timber species, and some small scale agriculture, causing forest degradation. Subsistence farmers may be evicted by commercial interests, such as cattle ranchers or soy cultivators. This pattern differs widely across continents.

Table 1. UNFCCC Best Estimates of the Direct Causes of Tropical Deforestation.

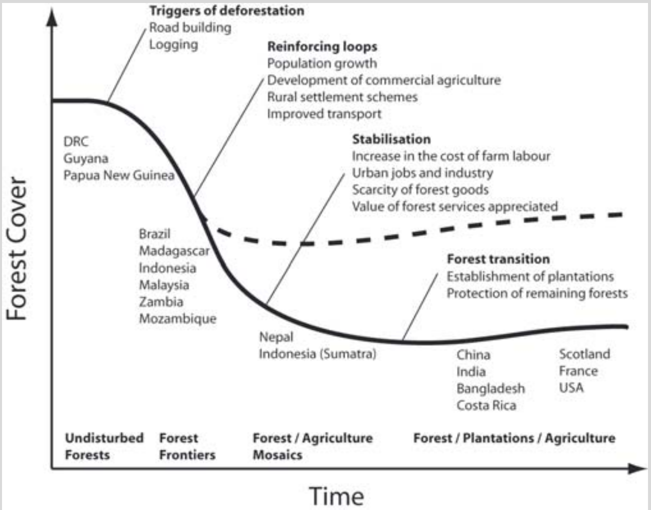
Approximately 129 000 km^2 are deforested each year, roughly equal to the area of England.[345], the causes being:

small scale agriculture/shifting cultivation	45%
commercial crops	20%
commercial wood extraction	15%
cattle ranching (large scale)	10%
fuelwood for own use	5%
fuelwood and charcoal (traded)	5%

As an economy develops, the pressures driving deforestation lessen. At the early stages of economic development, population and demand for agricultural land rise fast and forests are often cleared to make way for farms. Additionally, poor nations often try to increase exports of raw materials and encourage timber and other primary industries that cause deforestation. Profits from these industries create capital that is often invested in activities and transport infrastructure which fuel further deforestation. This leads to very rapid and accelerating deforestation. With further development, deforestation typically slows. This is because, as forest cover decreases, increasing scarcity and awareness of forest resources can prompt policies aimed at reducing its loss and also shortage of rural labour, which makes extensive agriculture less profitable, reduces deforestation rates.[346] For example, in China and India, the area of forest started to increase in the 1990s as a result of government policies that emphasised the value of the goods and services provided by the forest,[346] mainly from flood and soil erosion protection.

Figure 24. The Forest Transition

The dashed line shows the goal of international REDD Policy.



This change from deforestation to a stable or increasing forest cover is called the 'forest transition' and has occurred in nearly all nations with a GDP greater than $5,000 per person (Figure 24). Countries that do not develop as they deforest, but remain trapped in a cycle of poverty and subsistence farming (e.g., Ethiopia and Haiti), often continue to lose forest cover.[346] After the forest transition, forest cover remains stable (although often at low levels as in the UK at 4%) or gradually increases. However, this is often occurs through an expansion of secondary forests and plantations, sometimes with continued loss of old growth (primary) forests. Secondary forests and plantations typically do not have the same biodiversity and carbon storage benefits as primary forests.

Forests, Rainfall and Thresholds

Forests play a major role in the global water cycle. Up to half the precipitation falling on a typical tropical rainforest evaporates or transpires from trees (the loss of water vapour through the pores, stomata, in tree leaves). During the day, trees pump vast quantities of water into the atmosphere from the soil, and this leads to cooler, moister air in the vicinity and downwind of forests. Ocean winds can spread the moisture to create more rain, but because evaporation of water is stronger over the forest than the ocean, the lower pressure over coastal forests also sucks in moist air from the ocean generating wind to drive moisture further inland. This process repeats itself as the moisture is recycled in forests downwind, the forest acting as a pump to draw rain further towards the centre of a continental land mass.

Deforestation can disrupt this process and lead to changes in local rainfall and increases in temperature. Large scale deforestation is also predicted by climate models to have far-reaching effects on rainfall patterns, but the details are uncertain. At a regional scale, rainfall derived from forests can be critical to agricultural and other industries. For example, total rainfall in Haiti has fallen by as much as 40% as a result of deforestation reducing stream flow and irrigation capacity.[347] A major region of concern is the La Plata basin of Latin America, which generates 70% of the GDP of 5 countries and which is heavily dependent on rainfall carried downwind from the Amazon.

The interaction between deforestation, fire and climate change could trigger a widespread dieback, with parts of the forest moving into a self perpetuating cycle of more frequent fires and intense droughts leading to a shift to savanna-like vegetation.[348] While there are large uncertainties about where this threshold lies, the risk of crossing it increases greatly if deforestation rates exceed 20 to 30%. In addition to local impacts on the Amazon, there would be also global impacts on regulating services for carbon storage.[349]

Livelihoods and Poverty

Three hundred and fifty million people live in forests and 1.6 billion depend on them for their livelihoods and are often among the poorest, most marginalised people on the planet, both economically and politically. [350] The links between deforestation and poverty are complex. The rural poor in many developing countries depend on forests for fuel, food, medicine, grazing and fertile soils, and these resources are particularly important in times of stress, for example during droughts or war. However, rural poverty and population growth and the consequent need for land for subsistence farming are also a cause of deforestation. Studies have shown that improving agricultural productivity from existing farmed areas could play a substantial role in reducing deforestation.

Biodiversity, Soil Erosion and Flood Risk

Forest and other habitat loss is the major cause of biodiversity loss. Forests occupy approximately 31% of the Earth’s land surface and are estimated to contain more than half of terrestrial animal and plant species, with tropical rainforests the most diverse ecosystems in terms of plant and invertebrate species. They account for more than two thirds of net primary production (the conversion of solar energy into plant matter) on land.

Forests also modify the quantity of water in rivers, its quality and the evenness of flow, and can reduce the severity of floods as less water is retained or filtered by deforested habitats and there is an increase in soil erosion and sediments in water courses.,. In a similar manner, forests prevent soil erosion and landslides. Increasing sedimentation can impact coastal areas, in South East Asia for example, these sediments reduce coral growth in one of the most important areas for coral reef diversity.

Reducing Emissions from Deforestation and Degradation.

Around 16% of global CO_2 emissions are caused by deforestation, and halting it has been proposed as a cost-effective way of mitigating climate change. Parties to the UN Framework Convention on Climate Change agreed in 2007 that efforts to “Reduce Emissions from Deforestation and Degradation” (REDD) should play a role in climate change mitigation, partly because of co-benefits such as poverty reduction and biodiversity conservation. Funds for REDD have been provided by six nations, including the UK, under the Copenhagen Agreement, but the mechanisms for administrating these funds have not been agreed.

Under REDD, nations would be paid if they achieve a reduction in carbon emissions from deforestation. These payments could either be from a global fund, or as part of an international carbon market. A major cost is compensating for profits that would have been made if the land had been cleared for agriculture, estimated by the UK government’s ‘Eliasch Review’ at between $5-7 billion a year if deforestation were to be halved by 2030. If this was done as part of a carbon market, the costs would be higher – between $17-33 billion a year.[9]

However, other costs would be incurred under REDD, including those from improving governance, establishing land tenure and legal rights, and monitoring and forest protection. Where opportunity costs are very low (e.g. stopping clearance for subsistence farming in Africa) the set up and monitoring costs are likely to be very high, and lack of capacity and poor governance may make effective action impossible. The UN and World Bank have both set up funds to assist developing countries prepare for REDD. REDD would leave the national-level policies up to national governments, but could include safeguards promoting forest-peoples’ rights as well as biodiversity.

The COP16 in Cancun December 2010 included an agreement on REDD, “encourages developing country Parties to contribute to mitigation actions in the forest sector by undertaking the following activities, as deemed appropriate by each Party and in accordance with their respective capabilities and national circumstances: Reducing emissions from deforestation, reducing emissions from forest degradation, conservation of forest carbon stocks, sustainable management of forest and enhancement of forest carbon stocks”. However, many of the decisions were postponed, such as how the mechanism will be financed, until COP17 in Durban in December 2011. The agreement included safeguards on biodiversity and the rights of indigenous forest peoples, but the strength of such safeguards, and how they can be enforced, are an area of contention.

Soil degradation affects soil quality for agriculture and has implications for the urban environment, pollution and flooding. The biodiversity most important for agriculture is soil biodiversity: there is a high species diversity in soil,[351] but studies estimate that only 1% of soil organisms are currently known to science.[352] However, this diversity underpins key ecological processes that are essential for agriculture, such as soil formation, maintaining soil fertility, water cycle regulation and pest control.

Soil erosion is a natural process caused by wind and water, but is accelerated by human land use changes such as the removal of semi-natural vegetation, agricultural practices that damage soil structure and the disruption of other ecological processes can remove fertile soil that took hundreds of years to form.[353] Soil eroded by water not only results in the loss of productive soil and nutrients in the immediate vicinity, but can also silt up river channels damaging their biological quality, as well as transferring pesticides and nutrients.

These impacts can increase flood risk, reduce the lifetime and increase maintenance costs for built infrastructure, such as harbours, reservoirs and water treatment plants, as well as affecting fisheries and other human uses of aquatic systems. Since soil formation is slow, at an average rate of less than 1 cm soil depth per century, loss beyond this rate results in soil loss becoming an irreversible process, at least over human timescales, and the soils and the benefits flowing from the uses of soil will be reduced or cease entirely.

Soil quality in all UK ecosystems has been degraded by human activities over the last fifty years to some extent, primarily by atmospheric pollution and some by poor management such soil sealing or soil erosion. The consequences are reflected in the capacity of soils to regulate, buffer and transform chemical substances, including the capacity to store carbon, cycle nutrients and purify water. Soil organic matter is major terrestrial sink for carbon, with peat soils having the highest concentration of organic matter, and there are substantial losses of soil carbon to the atmosphere as a result of land use changes, such as deforestation (Box 23).

A3 Water

The water or hydrological cycle is a biogeochemical cycle, a pathway by which a chemical element or molecule moves through both biological and physical components of the planetary system. The flow of water over and beneath the Earth is a key component of the cycling of other biogeochemicals, with the salinity of the oceans derived from erosion and transport of dissolved salts from the land. The global water cycle is produced by water exchange between the atmosphere, the land, and the oceans, and its main components are precipitation on the land and the oceans, evaporation from the land and the oceans driven by the heat of the sun, and runoff from the land to the oceans.

The main processes of this cycle include: precipitation; formation of snow cover; snow metamorphosis and formation of ice; melting of snow and ice; interception of precipitation by vegetation cover and storage in land surface depressions; infiltration of water into soil and vertical transfer of soil moisture; evapotranspiration (from the leaves of plants); recharge of groundwater and ground flow; river runoff generation; and, movement of water in river channel systems.

The role of different processes in the water cycle and their description depends on the chosen spatial-temporal scales. These processes move water on, above and below the surface of earth but these processes are subject to increasing human impacts.[354] The total amount of water on the planet is unaffected by human impacts, an estimated 1.3 trillion cubic kilometres of water which 1% is fresh water, but human interactions with natural processes can change the chemistry, usefulness or availability of water.

Freshwater Systems

The key component of the terrestrial water cycle is generation of river runoff and movement of water in the river networks. The land surface of a river catchment area (the streams, lakes and all the land that drains into a river), together with the underlying geology, influences the quantity and quality of surface water and groundwater. Land use and water resources are intimately connected, and how land surface (for example agriculture practices) or subsurface (such as mineral extraction) is managed can directly and indirectly impact water resources. For example, the removal of hedgerows, ponds and woodlands or the draining of wetlands can increase surface run off and the risk of flooding, contribute to erratic river flows and reduce water quality through transfer of sediments, nutrients and pollutants. Removal of forested areas can have substantial impacts on the local water cycle of areas, including lowering of precipitation levels (Box 24). Conversely, unsuitable afforestation in arid areas can reduce water availability and may increase environmental degradation.[355]

The benefits of water provision to economic productivity are often accompanied by impairment to ecosystems and biodiversity, with potentially serious and unquantified costs.[356] Along with competing and growing demand from domestic, commercial, agricultural and industrial users, water is required to maintain functioning ecosystems and environmental flow requirements.[357] Freshwater ecosystems provide various services, including cleaning water, preventing floods, providing energy and regulating freshwater ecosystems. Sufficient environmental flow is critical to support freshwater ecosystems and to maintain appropriate water table levels in terrestrial ecosystems. Estimates suggest that at least 10,000 to 20,000 freshwater species are extinct or at risk globally. [358]

Due to human activities, the natural hydrological cycle of most river basins is highly modified. Only a small fraction of rivers globally remain unaffected by humans.[356] Dam construction and other infrastructure may regulate flows, however, excessive drainage of land or creation of impermeable surfaces can reduce capacity to buffer flows leading to increasing ‘peakiness’ of spates interspersed with low flows. The main stream flow regulation methods are construction of dams, levees, barrages, river channelization (in concrete culverts), dredging of river channels and weirs, which provide water accumulation, decreasing flood flow, and increasing low flow.

More than 75% of European Catchment areas are subject to multiple pressures and have been heavily modified, resulting in biodiversity losses.[359] Over 80% of lowland rivers and 60% of upland rivers have been modified in England. These changes typically result in habitat loss and changes to river flow that have impacts on freshwater biodiversity and water quality. Because some aquatic plants and microbes take up excess nitrogen from the water at higher rates than others, loss of key species or groups of microbes can also lead to reductions in water quality.[360]

Impacts of Pressures on Freshwater Systems

In dry regions, evaporation losses from the reservoir water surface may be so large that they seriously compromise any potential gains for human uses and water abstraction for agriculture irrigation can have considerable effects on the hydrological cycle of an area. For example, as a result of the rivers feeding it being diverted for irrigation since the 1960s, the inland Aral Sea shrunk to less than a tenth of its original volume, has split in two and become too saline to support most flora and fauna. Within fifty years, it has gone from supporting a substantial fishing industry to being mostly a desert covered with salt and toxic chemicals, resulting from weapons testing, industrial waste and pesticides runoff, which are picked up and carried away by the wind as toxic dust and spread to the surrounding area causing substantial human health impacts.[361]

Excessive abstraction can cause the groundwater table to drop steeply, and this may reduce the surface runoff and lower the level of flows in rivers, a significant issue in south-east England. Lower river flows often leads to a worsening water quality, as there is less dilution of pollutants, such as the nitrates and phosphates in treated sewage effluent. However, there have been significant reductions in discharges of organic wastes, levels of nitrates and phosphates in freshwaters and indications of improved water quality in most EU countries, including the UK, with positive trends for some aspects of freshwater biodiversity.

Invasive alien species are a significant problem in freshwater systems, which is likely to increase in coming decades in the UK (POSTnote 303). For example, freshwater bodies and their wildlife can be affected by:

- Floating plants such as floating pennywort that prevent light from penetrating the water body, killing plants deeper in the water and increasing the flood risk.
- Terrestrial plants such as giant hogweed and Himalayan balsam that grow along river banks preventing access, increasing flood risk and eroding river banks.
- Non-native crayfish carrying the crayfish plague which is lethal to the endangered native crayfish species, preying on native invertebrates and eroding riverbanks.
- Topmouth gudgeon and other non-native fish carrying parasites that threaten native fish species.

The most significant distortions of the water cycle are observed in urbanized areas. The replacement of natural land cover by the urban impermeable surface causes great reductions in infiltration into the land surfaces. The rainfall runoff from urbanized areas is mainly generated as overland flow and reaches the river drainage system very quickly causing rapid changes in the volume and velocity flow during high rainfall periods increasing flood risks (POSTnote 289).

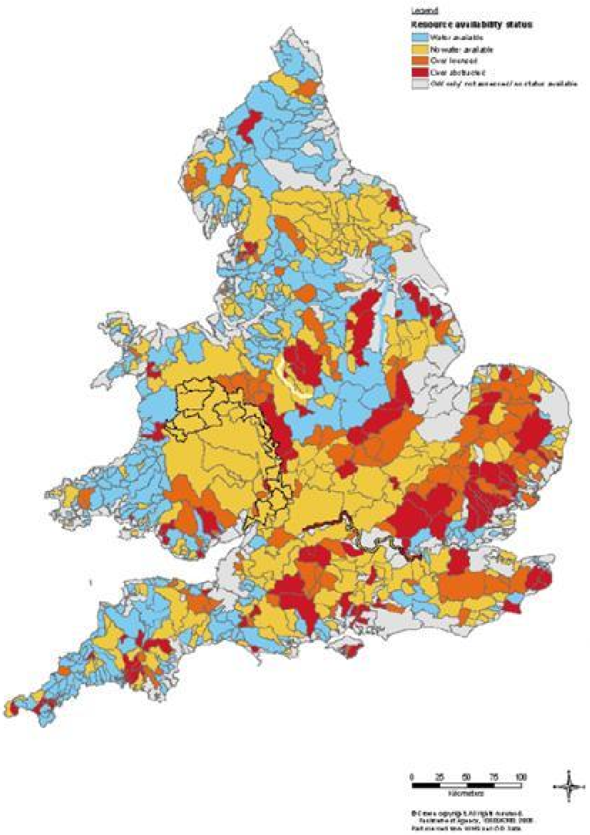
Water management systems are based around the concept that natural systems fluctuate within an unchanging envelope of variability (stationarity). This has led to decisions such as planning urban drainage systems to the specification of 1 in 30 year events. However, climate change will strongly influence future levels of fresh water supply through changes in means and extremes of precipitation, evapotranspiration and flow rates of rivers. Changes that have already occurred in climate have already altered systems beyond variables that built systems were designed to cope with. A warmer atmosphere increases humidity affecting precipitation rates and possible flood risk.[362] In the case of urban drainage, what were 1 in 30 year events have already become 1in 10 year events.[363]

EU and UK Water Use

Globally, humans appropriate more than 50 % of all renewable and accessible freshwater. By comparison, in Europe, only a relatively small proportion (13%) of its renewable freshwater resource is abstracted. In the EU as a whole, energy production accounts for 44 % of total water abstraction and for 50% in the UK, primarily for cooling water and hydropower. In the EU, 24% of abstracted water is used in agriculture, 21 % for public water supply and 11 % for industrial purposes. Some water abstraction is conservative, for example, water meadows that return abstracted water almost directly back to river systems, while other uses are consumptive and water is not returned, or may be returned remote from the point of abstraction and in a different condition (i.e. power station intakes).

However, in specific areas of Europe, including parts of the UK (Figure 25), the balance between water demand and availability has reached a critical level, the result of over-abstraction and prolonged periods of low rainfall or drought. In Europe, reduced river flows, lowered lake and groundwater levels, and the drying up of wetlands are widely reported, alongside detrimental impacts on freshwater ecosystems, including fish and bird life.[364] In the UK, the lower effective rainfall (rainfall minus evaporation) and high population density in

Figure 25: Catchment water resource availability status in England and Wales[365]



South East England means that net water abstraction may be approaching environmental limits in many areas.[366]

Overall, the amount of water put into public water supply in the UK declined between 1990 and 2008 from 20 billion (10^9) litres per day to about 17 billion litres per day, with the largest declines in England and Wales. A third of water abstracted in the UK is used for public supply and 52% of this is for household use. Government targets for the UK aim to reduce current per capita consumption of 150 to 180 litres per person per day (l/p/d) to 130 l/p/d by 2030. The UK person's average annual water use is 700 litres for drinking, and 60,000 litres for household and garden use, but the 'embedded' water used in the production food, products and services (for example, production of a single 300mm silicon computer chip requires 8,622 litres of deionised water) can amount to more than one million litres per person per year.[366]

The UK is the sixth largest net importer of embedded water in the world. Because the quality of groundwater is mostly far better than that of surface water, and its temperature is relatively constant, large volumes of groundwater are extracted for domestic and industrial use in different regions of the world. Agriculture is globally one of the greatest consumers of water and major shortages of water may have a major effect on food production. Substantial areas of the world's primary food-producing regions are being irrigated with retreating, unsustainable groundwater resources raising questions about food security.

In coastal areas, excessive extraction of groundwater also leads to seawater intrusion destroying the agricultural productivity of soil. Excessive irrigation can also cause saline intrusion caused by water soaking through the soil level adding to the ground water below. This causes the water table to rise, bringing dissolved salts to the surface. As the irrigated area dries, the salt remains, lowering the productivity of soils as occurred in the Murray Basin in Australia. Agricultural irrigation in the UK only amounts to about 1% of total abstraction, although it can have significant impacts as it is concentrated in the driest regions and at the driest months of the year.[366]

Impacts on Marine Ecosystems

Coastal and marine ecosystems provide a range of services including defence against rising sea levels, oxygen production, nutrient cycles, carbon sequestration, food and bioremediation of waste and pollutants. Modelling of the global oceans indicates that no area is unaffected by human influence and that 41% is strongly affected by multiple impacts from human activities, with the English Channel and North Sea being among the most heavily impacted areas.[367] Available data suggest significant and ongoing biodiversity loss in all European Seas, however, the data to document the extent and severity is incomplete, especially in offshore waters. The two largest human induced impacts on marine biodiversity are fishing and climate change, including ocean acidification.

The ecological impacts from fishing depend on the type, location and intensity of fishing. Impacts include direct removal of target species, bycatch of other marine organisms and damage to habitats caused by fishing gear that is dragged across the seabed. Marine ecosystems in UK seas have been extensively modified through selective removal of fish species, such as cod, and a reduction in the total biomass of larger species and individuals.[368] At least 45% of assessed European exploited marine fish stocks are outside safe biological limits. The EU was largely self sufficient in the supply of marine fish for human consumption until 1997, but domestic supply levels fell by over 50% by 2007, increasing reliance on exports.[369]

Biodiversity in areas with the lowest rates of natural disturbance is most negatively affected by human activities that disrupt the seafloor.[370] Muddy sediments tend to occur in locations with low disturbance and have lower rates of biodiversity recovery from physical impacts. Biodiversity on coarser sediments in areas with high levels of natural disturbance, such as strong currents or wave action, tend to have higher rates of recovery. Fragile seabed habitats, such as horse mussel beds, maerl and sea grass beds are not only prone to physical disturbance, but are also at risk from the diffuse pollution impacts of suspended sediments, nutrient enrichment and toxic chemicals.

Coastal waters are also subject to invasive alien species, with little surveillance in the marine system and limited control methods available for use. Non-native marine species can arrive in ballast water or through hull fouling. International Maritime Organisation (IMO) ballast water management guidelines (coming into force by 2016) require ships to clean ballast water to remove unwanted organisms. The IMO is also reviewing alternatives to the traditional chemicals used to prevent hull fouling that were banned due to their adverse effects on wildlife. Chinese mitten crabs arrived in the UK in ballast water. These increase flood risk by burrowing into and eroding estuarine banks (POSTnote 303).

Mean temperature in European continental shelf seas is rising faster than on adjacent land masses. Warming seas have led to warmer water species moving northwards.[371] Whether cold water communities are also shifting northwards or occupying a reduced habitat area is unclear. About 40% of CO_2 released into the atmosphere since the industrial revolution has been absorbed by the oceans. Consequences of this may include the loss of some marine species, habitats and ecosystems especially coral reefs and species in polar regions. Organisms with structures made from calcium carbonate ($CaCO_3$) are particularly vulnerable.

Climate change and ocean acidification will occur simultaneously. The impacts of warming and acidification together may be more severe than their impacts in isolation (POSTnote 343). Although they cover just 1.2% of the world’s continental shelves, it estimated that between 500 milion and more than one billion people rely on coral reefs as a food. They also support between one and three million species, including approximately 25% of all marine fish species.[372]

Wetlands are amongst the most productive and biologically diverse terrestrial habitats, and amongst the most vulnerable to drivers of environmental change, particularly coastal wetlands. Globally, over the last 20 years about 30 to 50% of the area of major coastal environments have been degraded, largely as a result of human use and development pressures.[373] As well as being nursery areas for commercial important marine species, coastal wetland areas are critical nutrient sinks that reduce the amount of nutrients entering marine systems.

In coastal waters, the build up of phosphorus and nitrogen, mainly through run-off from cropland and sewage pollution, stimulates the growth of algae and some forms of bacteria affecting water quality to the extent ‘dead zones’ form in some coastal areas. In these zones, the decomposition of algae removes the oxygen dissolved in the seawater leaving large areas virtually devoid of marine life. The number of reported dead zones has been roughly doubling every ten years since the 1960s, and by 2007 had reached around 500.[374]

Marine litter is another form of pollution that directly impacts biodiversity through physical entanglement and ingestion, including marine top predators. In particular, micro-plastic pollution may be vectors in transporting persistent and toxic substances that introduce toxic chemicals into the food chain.[375] Management of the impacts of marine litter have been included in the Marine Strategy Framework Directive.

A4 The Atmospheric Environment

The composition of the Earth's atmosphere has been changed by human activity and some of these changes have been harmful to human health and ecosystems including ozone depletion in the stratosphere, acid rain, photochemical smog and climate change. The composition of the atmosphere is changing at a rate that is significantly impacting a range of biogeochemical processes, including the carbon, hydrological and nitrogen cycles.

International agreements to regulate emission of sulphates, one of the major causes of acid rain, and to halt the production of chlorofluorocarbons (CFCs), the main cause of ozone depletion in stratosphere, have been successful in addressing these problems. The Montreal Protocol, agreed in 1987 in response to the dramatic depletion of the Ozone layer over the Antarctic, has led to the complete cessation of CFC production globally in 2010. However, this regulation does not apply to existing stocks of CFCs, which continue to be recycled and emitted to the atmosphere.

The Sulphur Emissions Reduction Protocol, under the Convention on Long-Range Transboundary Air Pollution, has led to the regulation of coal burning power stations to fit flue gas desulphurisation units to remove sulphur containing gases from the gases emitted. Emissions of oxides of nitrogen, which can be converted to nitric acid in atmosphere, continue to cause acid rain with subsequent impacts on forests, freshwaters and soils.

Air Quality

Ambient air quality, the condition of the air in the outdoor environment, directly affects the human wellbeing and ecosystems. Air pollution from major sources such as transport, power generation and industry are heavily regulated in developed countries, but continue to be a major impact on human health globally. The most serious effects on health occur at the greatest frequency at the highest levels of exposure to air pollutants.

However, the air pollutants particulate matter and ozone have no exposure threshold levels below which adverse human health effects do not occur, and are responsible for a significant number of premature mortalities, particularly in urban areas. Poor air quality reduces the life expectancy of everyone in the UK by an average of seven to eight months, and up to 50,000 people a year may die prematurely because of it. Air pollution also causes damage to UK ecosystems.[376]

Ambient Air quality has improved in the UK since the 1950s as a result of national and European regulation (POSTnote 272), but diffuse sources of atmospheric pollution remain a challenge, particularly transport sources. The Environmental Audit Committee recent report on air quality recommended that air quality should be a higher priority for the UK government. This included raising the priority attached to air quality in all government departments and provide better guidance on including air quality impacts in policy appraisals (Annex D). [377]

Critical loads are used to specify the maximum rates of oxides of sulphur and nitrogen emissions permissible, while avoiding adverse effects from acidification of soils and/or freshwater systems; or from eutrophication, where excess nutrient nitrogen leads to the deterioration of ecosystems and loss of biodiversity. Critical levels are used to define excess ground level ozone concentrations at which crop yields are reduced and forestry impacted. These are transboundary pollutants that can be transported long distances from where they are emitted. By contrast, ammonia is deposited close to where it is emitted, usually through agricultural activities, but is an increasingly important pollutant. Levels of oxides of nitrogen, particulate matter, ammonia and ozone remain a matter of concern in the UK (POSTnote 272).

Climate Change

The earth is habitable only because of the greenhouse effect. Without it, global average temperature would be around minus 18°C. The major greenhouse gases are water vapour, carbon dioxide (CO_2), and to a lesser extent methane. The United Nations Intergovernmental Panel on Climate Change (IPCC) has concluded that most of the observed increase in global average temperatures since the mid-20th century is very likely (more than 90% certain) to result from the observed increase in human caused greenhouse gases.[378]

Carbon dioxide (CO_2) is considered to be a major cause of global warming, contributing approximately 70% towards total greenhouse gas (GHG) emissions. Levels of atmospheric CO_2 alone have reached 387parts per million (ppm, referring to the ratio of the number of carbon dioxide molecules in one million molecules of air) in 2008; up from a pre-industrial concentration of 280ppm; [379] equating to around 800 gigatonnes (Gt) in the global atmosphere, and is rising by around 4 Gt/y.[380] In 2005, the global average concentration of methane was 1,774 parts per billion (ppb), up from the pre-industrial (1750) concentration of 715ppb.

It is very likely that the current atmospheric concentrations of CO_2 and methane exceed by far the natural range of the past 650,000 years. Data indicate that CO_2 varied within a range of 180 to 300ppm and methane within 320 to 790ppb over this period. Global levels of GHGs, usually expressed in terms of CO_2 equivalent (CO_2e), have risen to around 430ppm. The main sources of human GHG emissions globally are the burning of fossil fuels for electricity generation, transport, industry and household use. Other major sources include deforestation, agriculture and the landfilling of waste.

The uncertainties in climate science predictions, the requirements to mitigate greenhouse gas emissions and the need to adapt to the impacts of climate change have been extensively reviewed elsewhere.[381] If emissions are maintained at levels below 450ppm, there is 50% probability mean global temperatures on Earth will rise by no more than 2˚C. Atmospheric CO_2 concentrations are currently increasing by about 20 ppm per decade.[382] Regardless of whether or not international agreement can be reached to achieve this level of mitigation, most of the changes that will happen over the next 30 to 40 years have already been determined by past and present emissions of greenhouse gases. The average global air temperature has risen by 0.7 to 0.8 ºC since pre-industrial times.[383] Over the last three decades, temperature increase has already had a discernable influence at the global scale on natural systems. (POSTnote 341). The UK will experience warmer and wetter winters and hotter and drier summers. The incidence of severe weather events will increase and sea level rise and higher storm surges will increase the risk of coastal erosion and flooding (Postnote 342).

Climate change interacts with other environmental issues, such as biodiversity and patterns of crop and animal disease in the UK (POSTnotes 307 and 341). It will also increase the risk of changes to ecosystem functioning. For example, changes in temperature, drought events and precipitation intensity could increase the risk of soil erosion, decrease organic carbon stocks in soils and increase CO_2 emissions from them. The impacts of climate change will vary across the country, the south and east experiencing the highest levels of water stress, with the risk of flood events increasing in many river basins, particularly in the spring and winter. The UK is also likely to experience knock-on effects from the impacts of climate change in other parts of the world, such as changes in the security and the supply of food and raw materials.[384]

Annex B Natural Capital Indicators

B1 Natural Capital and National Accounts

The System of National Accounts (SNA) consists of an integrated set of macroeconomic accounts, balance sheets and tables based on internationally-agreed concepts, definitions, classifications and accounting rules to provide quantitative information about a national economy. It acts as an accounting framework within which economic (and some social and environmental) statistics can be compiled and used for purposes of economic analysis, decision-taking and policy-making and to compare with the performance of other countries, including variables such as GDP, production, investment and consumption.

There are two main categories: the stock account of assets and the flow accounts of goods and services arising from those stocks. Key aggregate statistics of these categories are used as indicators of national economic activity, such as gross domestic product (GDP). In general, the value of goods and services within this system is based on their commercial market value. From this, their contribution to increasing GDP can be calculated. It is widely recognised that the current SNA does not reflect the full costs and benefits to society of economic progress. GDP is only a measure of economic activity, not human well-being or wealth and, as such, it does not include the value of non-market goods and services arising from natural capital such as ecosystem services, although they underpin economic activity.

The depletion of natural capital increases GDP through the economic activity arising from its extraction and use, but does not reflect the environmental cost of this depletion. Mitigating the impacts of the depletion and degradation of natural capital, such as dealing with the impacts of pollution, can also increase GDP, while leading to an overall decrease in well-being.

Most governments regularly monitor the level of gross fixed capital formation (GFCF) as a measure of investment in infrastructure, but not levels of natural capital stocks. However, by ignoring scarcity in natural capital, the risks and probability of costs incurred through impacts on future economic productivity are not taken into account; nor are their implications for future security and human health and well-being.

Measuring Natural Capital Depletion

Those aspects of natural capital use or degradation that are not monitored are likely to be ignored in policy processes. This also results in a subsequent lack of public investment in natural capital stocks through conservation and land management policies. Environmental accounting attempts to incorporate measures of the depletion and degradation of environmental assets and functions into national accounts, with corresponding modifications of key economic indicators.

Resource functions, the supply of raw materials from the environment, are generally traded as market goods and are already in accounts. The use of, or impact on, ecosystems (service and sink functions), such as releasing pollutants into the environment to dispose of them, are not marketed and are therefore not straightforward to incorporate into national accounts.

This requires two types of accounting adjustment: one for the flow of benefits from natural capital and one for changes in levels of natural capital stock. The environmental costs and benefits of an economic activity can accrue to either social welfare or affect other economic sectors (in terms of both current and future risks). For example, flooding caused by changes in land use, such as the destruction of wetlands, will result in an increase in GDP due to both the agricultural production arising from the land conversion and the repairs necessary to infrastructure and domestic dwellings. The costs of flooding will accrue to the insurance industry, private individuals and to social welfare in terms of health impacts and new flood prevention measures.

Accounting adjustments also need to be classified in terms of whether they measure changes in wealth or income. The loss of natural capital stock, such as the loss of wetland, results in loss of wealth, as the future benefits arising from that natural capital are lost. Reductions in the flow of benefits from natural capital, such as flood regulation, are a negative change in income.

Natural capital accounts could be used in the same way that economic accounts are used to measure economic activity. This would entail identifying the different stocks of ecosystem elements, for example broad habitat types such as wetlands, the activities or pressures that affect them and the consequences for the flow of ecosystem service benefits. This can be done through representing the quantity and quality of natural capital stocks in physical units and through monetary valuation of the flow of services.

The UN System of Environmental and Economic Accounts (SEEA)

There has been reluctance to implement the SEEA at the international level, possibly because the methodologies are not regarded as robust enough by national financial ministries. The global application of environmental accounting conventions under SEEA would allow international comparisons of the environmental impacts of national economies and could be extended to include the maintenance of irreplaceable natural capital.

However, the methodologies for inventorising and valuing the majority of environmental benefits arising from natural capital, such as water quality, have yet to be fully developed. This is in terms of both developing physical indicators of the impacts of economic activity on the environment and the appropriate ways of incorporating the data into the accounts, through monetisation or other means.

In physical accounting approaches, different issues need to be measured and reported in the most appropriate units. For natural resource use, the most common units of measurement at present are mass units (kilograms), energy units (joules), area units (hectares) or units which reflect the negative environmental impacts of resource use on human health (for example, healthy life years or numbers of premature deaths). Further revision of the SEEA in 2013/2014 will clarify what is known and the data available, and further identify impacts where there is a paucity of scientific information. The development of a system of ecosystem service benefit accounting is a specific issue to be addressed (Box 7).

The SEEA will be revised in line with the recommendations of the UN Committee of Experts on Environmental Accounting and will look at experimental green accounting methodologies including ecosystem accounting and classification schemes for ecosystem services and ecosystem assets, such as CICES (Section 3.5). This revision is intended to establish international standards for environmental accounting. According to the SEEA, natural capital is considered to comprise three principal categories:

- Resource functions, which cover natural resources drawn into the economy to be converted into goods and services, such as timber. This includes non-renewable resources such as fossil fuels and metal ores as well as renewable resources.
- Sink functions which absorb waste products of production, such as pollutant gases, that are released into the air, water or buried in the ground.
- Service functions, which sustain the conditions for life, such as air to breathe, water to drink or soil to grow crops, they also include cultural and recreational services.

The resource function of ecosystems (such as timber), can be calculated according to conventional economic rules and some sink functions can be valued through the costs of restoring impacted ecosystems. However, the valuation methodology remains unclear for loss of service functions (such as the degradation of a forest supplying timber).

National Accounting Matrix Including Environmental Accounts (NAMEA)

To determine and monitor the impact of the economy on the environment, satellite environmental accounts to the main National Accounts are advocated by the UN. The SEEA provides the potential for international comparability in environmental national accounting. It encompasses a wide range of activities and measures, including natural resource balances, emissions, waste and environmental expenditure accounts.

In line with UN Statistical Office requirements (Box 7), the statistical office of the European Union (Eurostat) requires all EU member states to construct a National Accounting Matrix including Environmental Accounts (NAMEA). The NAMEA concept contains a summary of environmental indicators. The environmental accounts are reported in physical and monetary units, and focus on presenting information on material inputs of natural resources (particularly energy resources) and outputs of emissions (pollution and waste materials) at a level of sectoral detail consistent with national economic accounts.

The UK has already adopted the Eurostat guidelines in reporting economic-environmental accounts in the NAMEA format at the national level. For example, it report is the use of different types of energy and generation of a number of air pollutants by 76 production sectors and by households. The UK environmental accounts are used to inform sustainable development policy, to model impacts of fiscal or monetary measures and to evaluate the environmental impacts of different sectors of the economy, and to assess the degree to which economic growth is degrading the environment for future generations.

B2 Adjusting Economic Indicators for Environmental Impacts

There have been several international projects considering whether economic indicators could incorporate measures of environmental impacts through the development of appropriate aggregate indicators. For example, recent initiatives have considered whether GDP could be broadened to encompass other aspects of well-being beyond economic activity. These include the EU "Beyond GDP" initiative, UNEP's "Green Economy initiative", and the International Commission on the Measurement of Economic Performance and Social Progress (CMEPSP), while the OECD has a group, "Measuring Progress", to consider revising sustainable development and well-being indicators.

Empirical studies to compute environmentally-adjusted net domestic product (eaNDP), or "Green GDP", taking into account the consumption of natural capital pre-date the establishment of SEEA. NDP already takes account of depreciation in human capital and is arguably a better measure of well-being than GDP. The possibilities for taking account the consumption of natural capital within the SEEA framework have been considered at some length by previous initiatives, such as the "Green Stamp approach" (Valuing Damages for Green Accounting Purposes: The GARP II Approach) funded by the EU. One possibility would be to assess the value of national natural capital stocks without market values and add these to GDP to give an Inclusive Domestic Product (IDP), to better reflect well-being. The depreciation of natural capital stocks, including the full environmental cost of imported goods and services, could then be calculated as a separate aggregate indicator, the "Full Costs of Goods and Services" (FCGS). This requires valuing in monetary terms each indicator of natural capital depreciation.

The ratio of FCGS to GDP or IDP would then be used to reflect the over-use of environmental assets as inputs to economic production and environmental degradation of resources. A current example of such an approach is the "Green Accounting" for the Indian States Project (GAISP), in which ecosystem services with no market values are assigned an economic value and added to GDP. Ecosystem degradation and depletion costs are then deducted from this value, and a ratio between the increased GDP and the environmental losses is derived.

Alternatively, the costs of restoring or maintaining ecosystem assets at levels to deliver ecosystem service benefits (domestic and imported) at required levels can be used as an estimation of ecosystem capital depreciation to adjust GDP, or to derive Adjusted Disposable National Income (Section 3.9). If the full costs of maintaining ecosystem service are not met, the adjusted GDP will fall in both cases.[385]

The inclusion of some measure of environmental degradation within GDP has been the subject of criticism. It is argued that it is difficult to calculate a robust figure for the costs of environmental degradation (quantitatively or methodologically) and that Green GDP just 'charges' GDP for the depletion of damage to environmental resources rather than providing an assessment of how far an economy is from environmental sustainability targets.[386]

It also makes the implicit assumption that natural capital can be substituted by man-made capital, such as technology or other infrastructure, which is in general not possible. In addition, while being a good indicator of economic productivity, GDP is a poor indicator of well-being in general. Many economists advocate that any indicator of the impacts of an economy on the environment should be separate from GDP.

Adjusted Net Savings

One of the most prominent exponents of national green accounting is the World Bank. There are accepted systems within SEEA for estimating certain natural resources at the national level, such as energy, minerals, and forests, with well developed accounting rules for determining how much of these assets is being preserved for future generations.

Such measures of "genuine saving" identify the change in real wealth of an economy after due account is taken of the depreciation and depletion of natural resources and the investment of proceeds in other aspects of the economy. Drawing on the valuation methodologies in SEEA, the World Bank publishes "adjusted net" savings estimates annually in the World Development Indicators, which include:

- estimates of consumption of produced assets, which are deducted from standard national accounting measures of gross national savings;
- expenditure on education, which is added as investment in human capital;
- estimates of the depletion of a variety of natural resources, which are deducted to reflect the decline in asset values associated with their extraction and harvesting; and,

- global damages from carbon dioxide emissions, which are deducted.

This indicator excludes depletion costs of various natural resources such as soil, water, fish and genetic material and some degradation costs such solid wastes and water pollution. The outcomes of the indicator tend largely to reflect the consumption of produced assets and investment in education in developed countries, which are mainly sustainable according to this indicator, whereas developing countries that export natural resources are classed as unsustainable.

Hence the dependence of developed countries on natural resource imports, and in some cases on outsourcing hazardous industrial processes, is not taken into account, so that developed countries effectively import environmentally sustainability from less developed but resource rich countries. Critics also state that this indicator attempts to substitute natural capital with human capital (for example, expenditure on education).

B3 Environmental Aggregate Indicators

Measuring resource use and its environmental, economic and social impacts through appropriate indicators is a prerequisite for monitoring whether policies are increasing the likelihood of breaching environmental limits or progressing towards agreed environmental targets. Several aggregate indicators or summary environmental indicators based on different physical accounting methodologies (Box 24) have been proposed to adjust national economic accounts to reflect the over-use of environmental assets and environmental degradation of resources, and the consequent impacts on human well-being.

Five basic categories of natural resources need to be accounted for: biotic materials (from living organisms), abiotic materials (physical components of the environment), air (for combustion processes and a sink for wastes), water and land area. However, different measurement methodologies are required for each of these categories, posing difficulties for developing aggregate indicators.

There is an extensive academic literature discussing the feasibility and shortcomings of national aggregate environmental indicator approaches, the main points of contention being uncertainties in understanding the importance of the different components of natural resource systems so as to weight them or assign monetary values and how to assess the 'resilience' of natural resource systems and ecological risks (Chapter 5).

Suggested aggregate indicators include the World Bank's Adjusted Net Savings, the United Nations Development Programme (UNDP) Human Development Index (HDI), CMEPSP Dashboard of Sustainable Development Indicators, WWF's Living Planet Index, the Index of Sustainable Economic Well-being (ISEW) and the New Economic Foundation's (NEF) Happy Planet Index. All of these approaches have been shown to be valid communication and awareness-raising tools, but their robustness for national environmental accounting is questionable.

The 2009 European Commission communication, "GDP and Beyond: Measuring Progress in a Changing World",[387] stated that it intended to:

- develop a comprehensive environmental index which captures harm to or pressure on the environment in the European Union and to explore a comprehensive indicator of environment quality;
- continue to develop indicators of quality of life and well-being;
- aim to provide "near real-time" information for decision-making;
- provide more accurate reporting on distribution and inequalities;
- develop a Sustainable Development Scoreboard and step up efforts to identify thresholds for environmental sustainability; and,
- extend national accounts better to include environmental and social issues – specifically integrated environmental-economic accounting which requires monetary valuation of changes in environmental assets.

The 2007 EU "Beyond GDP" conference concluded that a basket of four indicators would be sufficient (Ecological Footprint, Human Appropriation of Net Primary Production (HANPP), Landscape Ecological Potential and Environmentally Weighted Material Consumption) to form part of a core set of headline macro-economic aggregates, alongside conventional GDP, National Income and Final Consumption. The European Environment Agency (EEA) has proposed additional ecosystem indicators relating to water and biodiversity should also be included in national environmental accounts.

Indicators for Natural Capital Stock

Ecological infrastructure is the natural capital required to deliver ecosystem services in a given region (section 2.3), including freshwater systems, grassland systems, forests, wetlands, soils, agricultural land and fisheries. A major component

Box 24 Measurement Methodologies for Aggregate Environmental Indicators

Ecological Footprinting

The ecological footprint methodology was developed in the 1990s, and since then has been regularly updated by the Global Footprint Network. On the basis of this methodology, the WWF has produced regular footprint studies as part of its annual Living Planet Report. The Ecological Footprint "measures how much of the regenerative capacity of the biosphere is used by activities". The amount of productive land and marine area required to support a given population at its current level of consumption and available resources with prevailing technologies is calculated. Food, fibre, energy, building materials, fresh water resources consumed as well as environmental processes required to absorb wastes and support infrastructure (including urban areas) are converted into a measure of land area called 'global hectares'.

At the end of a survey, these footprint values are categorised for carbon, food, housing, and goods and services as well as the total footprint number of Earths needed to sustain the world's population at that level of consumption. The methodology defines "biocapacity" as the capacity of the biosphere and its ability to provide biological resources and services useful to humanity. On the basis of this definition, it has been calculated that the global environmental footprint of humanity exceeds the Earth's "biocapacity" by 25%.

The total world ecological footprint is 2.7 global hectares per capita and the ecological reserve, or "biocapacity" - the amount of land available for production, is in deficit at 0.6 global hectares per capita (gha). The UK's average ecological footprint is 5.45 gha with variations between regions ranging from 4.80 gha (Wales) to 5.56 gha (eastern England).[388] This has proved to be an effective communication tool for environmental NGOs, and the concept is readily understood by the popular media and public.

Indicators such as the WWF Living Planet Index are designed to highlight the global inequity associated with the acquisition of the world's natural resources and show that richer, more developed, countries are the primary agents of environmental impacts. In addition, they highlight the likely future increase in impacts as developing countries aspire to the lifestyles of developed countries. Given the disparity in global wealth and resources, it is unlikely that there will be a global convergence towards lifestyles that are in line with global "biocapacity".[389]

The shortcomings of the methodology have been discussed at some length in academic literature[390] and by the recent International Commission on the Measurement of Economic Performance and Social Progress CMEPSP. The methodology is designed to estimate available resources in relation to consumption and production, rather than to encompass all environmental sustainability issues, and acts as indicator of a country's contribution to global unsustainability, rather than a country's sustainability. For example, densely populated countries will have low biocapacity and ecological deficits, regardless of individual rates of consumption, while sparsely occupied ones, such as Finland, enjoy a surplus. Thus, countries' ecological footprints are now compared with global "biocapacity" rather than with the country's own "biocapacity".[391]

Much of the increase in ecological footprints over time also appears to be driven by the increased level of carbon dioxide emissions rather than the "biocapacity" of fishing grounds, forest lands or crop lands, for which the methodology implies there is no global sustainability problem for natural capital in contrast with many other indicators.[392] The re-weighting of land areas according to their relative potential for agricultural productivity, to derive global hectares, also gives greater weight to cropland over other land uses and benefits, including ecosystem services other than food provisioning.

Life Cycle Assessment

Life Cycle Assessment (LCA), compares the range of environmental and impacts assignable to products and services to identify the least burdensome processes, to optimise the environmental performance of a single product or that of a company. The term 'life cycle' refers to the notion that for a holistic assessment all the steps in raw material production, manufacture, distribution, use and disposal (including all intervening transportation steps) need to be assessed. LCA should be conducted in accordance with the internationally agreed standard ISO 14040:2006. This describes the principles and framework for LCA which includes:

- definition of the goal and scope of the LCA;
- the life cycle inventory analysis (LCI) phase;
- the life cycle impact assessment (LCIA) phase;
- the life cycle interpretation phase; reporting and critical review of the LCA;
- limitations of the LCA;
- the relationship between the LCA phases; and,
- conditions for use of value choices and optional elements.

For example, the 'Carbon Footprint' methodology assesses greenhouse gas emissions throughout the complete supply chain of goods and services consumed in a region or country using LCA approaches. The 'Carbon Footprint' concept has been applied at the product level to provide information to consumers through a standardised labelling approach.[393] Indicators such as 'Water Footprints' and 'Carbon Footprints' are a sub-set of the data covered by an integrated LCA. However, the information intensive requirements of LCA mean that it is unlikely to be the basis of an aggregate indicator in itself although it can be used in combination with other approaches such Materials Flow Analysis and Footprinting.

Nonetheless, information from the LCA of a product or service can be used directly to inform policies, for example, 'product roadmapping' to:

- identify the impacts that occur across each product's life cycle;
- identify existing actions being taken to address those impacts; and,
- develop and implement a voluntary action plan to address any gaps;

Defra is currently developing roadmaps with a range of government and private sector collaborators to reduce the environmental impacts of products in the four product groupings that account for 70-80% of all environmental impacts and 60% of consumer expenditure:

- food and drink (20-30% of impacts);
- housing, including buildings, construction and appliances (20-35% of impacts);
- passenger transport (15-35% of impacts); and,
- clothing (5-10% of impacts).

Materials Flow Analysis

Material Flow Accounting and Analysis (MFA) focuses on the use of different materials by human activities, based on national or international statistical data. The principal concept underlying the economy-wide MFA approach is a model of the relationship between the economy and the environment, in which the economy is an embedded subsystem of the environment and dependent on a constant throughput of materials and energy. Material flow-based indicators can be aggregated from the micro to the macro level and provide aggregate background information on composition and changes of the physical structure of socio-economic systems, and could be the basis of a set of aggregated indicators for resource use.

Raw materials, water and air are extracted from the natural system as inputs, transformed into products and finally re-transferred to the system as outputs (waste and emissions). For a consistent compilation of an economy-wide material flow account, it is necessary to define where the boundary between the economic and the environmental system is set, as only resources crossing this border will be accounted for, one boundary being between the economy and the environment, the other being the national boundaries between which materials and energy are imported and exported.

MFA calculates the domestic extraction of resources, as well as physical imports and exports. Biotic materials cover production from agriculture, forestry, fishery, and hunting; abiotic materials cover minerals (metal ores, industrial and construction minerals) and fossil fuels (coal, oil, gas and peat). The standardised methodologies for accounting and analysing material flows at the national level have been set out by Eurostat and OECD.

There are several problems with aggregate MFA based indicators. Aggregate material flow accounts (MFA) indicators are Direct Material Input (DMI), Domestic Material Consumption (DMC) and Total Material Consumption (TMC, including the hidden flows) which change very little over time. This is partly due to the high level of aggregation. Furthermore, the aggregate MFA indicators are closely linked to economic events, rather than policies. These two points restrict their value as policy indicators, a major issue being that negative environmental impacts cannot always be easily conveyed by quantitative information, as there are large differences in the environmental impacts for different resources or materials. A kilogram of sand does not have equal impacts as a kilogram of copper, or meat, or coal. The potential environmental impacts of the different materials or resources need to be conveyed in an indicator as well as the weight or volume of their use.

There have been several methodological attempts to combine environmental impacts and material flows, such as the development of an Environmentally Weighted Material Consumption (EMC) indicator for the EU and Environmental Loads for the UK Environment Agency. EMC is a weighted indicator of material consumption based on environmental impacts per kg of material over the life cycle.[394] The EMC combines a set of specific impact indicators (e.g. CO_2 emissions, land use) that are then aggregated using weighting factors. A range of environmental issues is included in the EMC, such as human-health, eco-toxicity impacts of certain materials, ozone depletion, eutrophication, and acidification.

MFA-based approaches have also been combined with MFA and LCA methodologies to define Material Input per Service Unit (MIPS), that illustrates the material inputs required along the whole life cycle of a product, from resource extraction and refining via manufacturing and trade and consumption and, finally, treatment and disposal. However, as this incorporates both direct and indirect material flows, it requires the compilation of an enormous amount of material input data at each stage of production.

Land Cover and Land Use Accounts

Land Cover Accounts are generally established from satellite images using a particular resolution. For example, the Land and Ecosystem Accounts (LEAC) is a method developed and used by the EEA to account for the interactions between nature and society on the basis of a detailed grid (1km x 1km) for land use and land cover changes within the European Union. It is based on CORINE (Co-ordination of Information on the Environment) land cover data, using satellite images in a 100m x 100m grid and its goal is to provide information on land cover and related land use changes. Within LEAC, ecosystem accounts incorporate material and energy stocks and flows with measures of ecosystem service benefit provision (Sections 3.8 and 3.9).The ultimate goal is to measure the resilience of natural capital, its services and maintenance costs (Figure 22).[395]

The latest CORINE land-cover inventory, for 2006, shows a continued expansion of artificial surfaces, such as urban sprawl and infrastructure development, at the expense of agricultural land. Extensive agricultural land is being converted to more intensive agriculture and in parts into forests.[396]

of integrating resilience and estimation of risks into ecosystems into any accounting framework valuing natural capital will be the production of physical accounts of these ecosystem assets. The depreciation of their stocks would allow the effect of environmental degradation on the flow of future services to be estimated.

The state of such ecosystems is usually assessed through indicators (measures that summarise complex data into simple, standardised and communicable figures, such as the farmland bird indicator). The UK currently publishes a set of “Sustainable Development” Indicators in Your Pocket (SDIYP), which covers indicators across 68 different fields, many pertaining to well-being, including bird populations, fish stock sustainability, river quality and air pollution impacts, crime, employment and worklessness, poverty, education and health, social justice and subjective well-being. There is general agreement that as wide an array of indicators of quantities and qualities of natural resources as possible should be included in any measurement or accounting system to overcome the shortcomings of any individual indicator. Any suite of environmental indicators will need to give quantitative information about the range of different factors that will affect human well-being.

The indicator sets for natural capital should provide information in variations of different stocks, (for example the status of stocks of commercially-exploited marine species), whether the stock is increasing or decreasing, and how far it is above critical thresholds. Loose definitions of natural capital could lead to double counting and undermine accounting systems. The Defra Natural

Value Programme is considering the issues around the development and use of an asset check for natural capital.[397]

Driving Force-Pressure-State-Impact-Response (DSPIR) Framework

The lack of data for relevant ecosystem service indicators is a major constraint to developing ecosystem accounts. Indicators for monitoring ecosystem services are an essential tool for communicating complex patterns and processes to decision-makers and for measuring the success of policies to maintain them or negative trends that need addressing.[398] Ecosystem service indicators would encompass many concepts and processes, raising difficulties in interpreting them, as well as relating only to certain aspects of biodiversity or ecosystems.

The “driving force-pressure-state-impact-response” (DSPIR) framework is a commonly used approach to structure and analyse indicators.[399] It aims to describe interactions between society and the natural environment:

- Driving forces are the societal, demographic, and economic changes such as population growth and development.
- Pressures are the consequence of these changes and include land use and emissions of greenhouse gases.
- State is the condition of the environment in terms of quality and quantity, such as the abundance and distribution of various species or number of water bodies reaching ‘good ecological status’ under Water Framework Directive criteria.
- Impacts are effects on ecosystems and human health resulting from adverse environmental conditions, such as the number of properties flooded annually or rates of loss of a habitat type such as wetlands, hay meadows or ancient woodland.
- Responses are the measures taken to address the drivers, pressures, state or impacts such as area of land under agri-environment schemes.

Indicators fit into one of these categories, although the majority of environmental indicators at present fall into the “state” and “impacts” categories.

Indicators and Environmental Limits

Environmental limits can be defined on the basis of pressure, state or impacts indicators:

- The level of an environmental pressure on natural resource system and related ecosystem services (such as the level of greenhouse gas emissions).
- The state of the natural resource and related ecosystem services (such as the presence or absence of various species of organisms).
- The level of a benefit provided by a natural resource system (such as flood protection).

Only the last of these would fall within the strict definition of an environmental limit (Chapter 2), but the pressures applied to natural resource systems and their resultant state will dictate the level of benefits provided. The DSPIR framework is intended to provide a map of environmental pathways that demonstrates how, for example, a change in water quality may be linked to changes in agricultural practices, such as the application of fertilisers. Ideally, there should be sufficient information to map each economic activity and the drivers, changes of state, impacts and effectiveness of mitigating responses. However, for the most part, there is insufficient research to determine the causal links between impacts and driving forces and so only associations can be inferred.

Ecosystem Indicators and Accounting

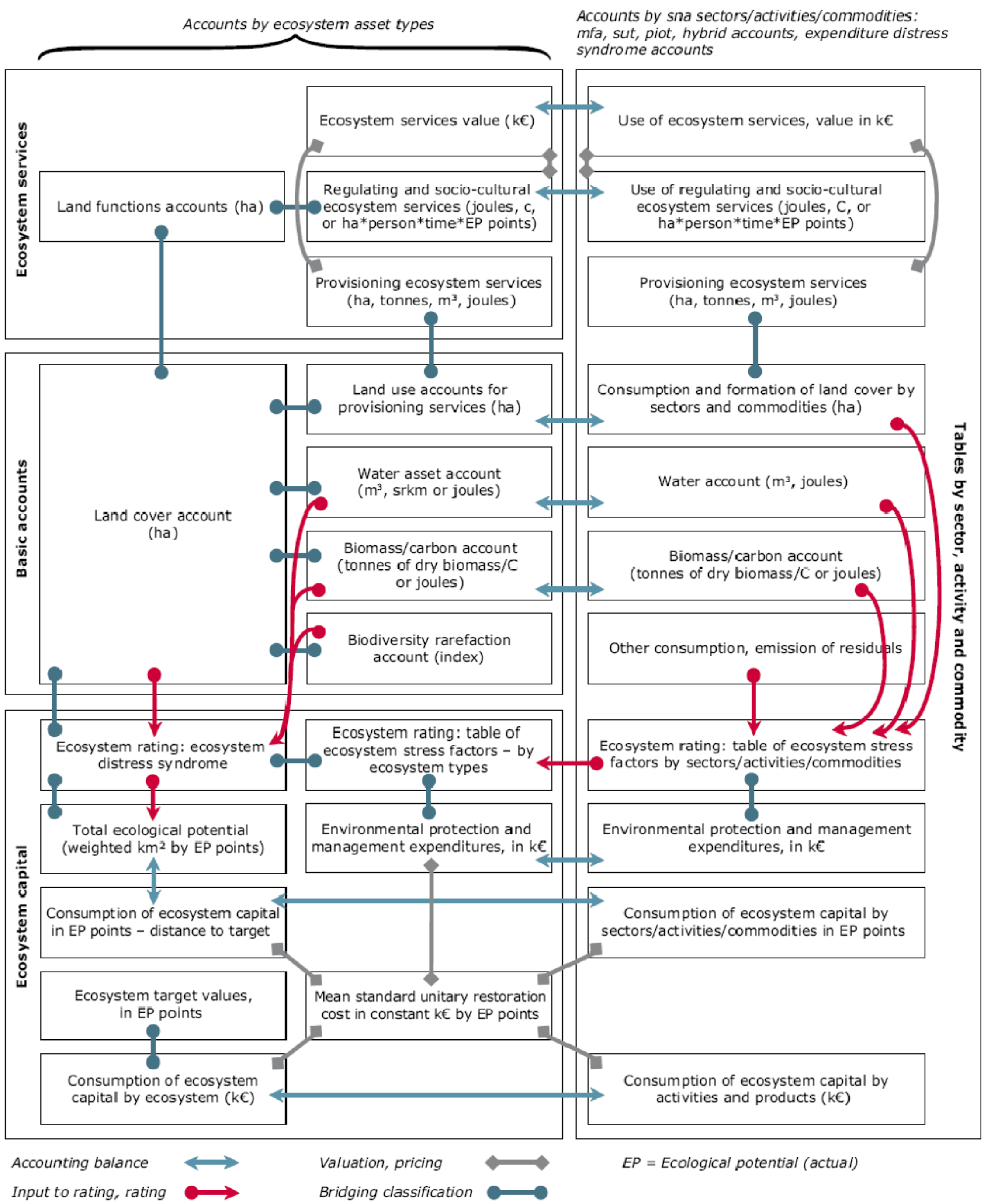
The European Ecosystem Assessment (EURECA) will assess the state of ecosystems in Europe in 2010 and their possible development beyond 2010. EURECA is intended to contribute to the next round of the Millennium Ecosystem Assessment as a sub-global assessment. It will include an evaluation of the stocks, flows and value of selected ecosystem goods and services under different policy-relevant scenarios, a comprehensive spatial analysis of ecosystems and an assessment of vulnerability to socio-economic and climate change.

Land accounting has been established on the basis of land-cover change detection for Europe. [400] There will also be in-depth studies of selected issues and areas including “Fish Behind the Net”, “Food for Thought”, “Natura2morrow” and “Peak Performance” to develop indicators in specific areas. EURECA will contribute to the review of the 6th Environment Action Programme and produce reports on the implementation of the EU Biodiversity Action plan, together with the “Streamlining European 2010 Biodiversity Indicators” (SEBI) (POSTnote 312).

Ecosystem Service Indicators

It remains a matter of contention whether the full costs of maintaining ecosystem services can be represented within an environmental accounting framework. The EEA has suggested a structure for national ecosystem accounting (Figure 26), which can be broken down into three major components: [401]

Figure 26: The Framework for Ecosystem Accounting and the Calculation of the Full Cost of Ecosystem Goods and Services Suggested by the EEA. [401]



- A set of basic accounts describing the important stocks and flows that constitute natural capital and its uses. These describe the quantity of the different ecosystems, measured in terms such as area of habitat, and the use of these assets by different economic sectors.
- A second set of accounts describing the condition of the ecosystem capital base, which documents the status and integrity of ecosystems.
- A third set of accounts that document the output of ecosystem services their uses and values.

The EEA suggests an expanded portofolio of indicators arising from these accounts, which can be used to describe total ecological potential (TEP). This is intended to describe the capacity of natural resource systems to sustain biodiversity and provide ecosystem services based on the measurement of the density of high biodiversity land-cover types at different spatial scales, and the fragmentation of such areas by roads and other infrastructure. However, the EEA accepts, there are still many data gaps and scientific uncertainties and the construction of a complete set of ecosystem accounts remains a challenge (Section 3.3).[401]

A key scientific challenge is how to define and measure complex biological and environmental processes and how to link these to levels of ecosystem service benefit provision. This is critical to quantifying ecosystem services and understanding the role of biodiversity in ecological processes underpinning service provision at relevant spatial and temporal scales. This evidence can be used to predict likely changes and responses of ecosystems to anthropogenic pressures and to design policy mitigation measures.

Building on the EEA proposals, the CICES classification aims to describe the links between ecological assets and processes and the benefits that flow from these, but places a value only on the final outputs that directly or indirectly contribute to human well-being, to avoid 'double-counting' of benefits (3.3). It also consists of generic categories in a nested hierarchy to accommodate benefits at different scales of concern and new benefits as data become available (Figure 27).

A nested hierarchy allows food production to have a different status from less essential benefits such as ornamental plant resources. These classifications can also be linked to other economic classifications for products and services although such cross-tabulation is likely to be complex. To develop accounts that are able to link changes in ecosystem structures and processes to their economic consequences, there will also be to cross-tabulation between the CICES classification and ecosystem service indicators. [402]

Defining Suitable Indicators

The purpose of indicators determines the type needed to address a problem and the spatial scale of application. For example, preserving biodiversity at a level to sustain ecosystem services is likely to be very different from that required to meet the 2010 halting biodiversity loss target, and will require a different array of indicators. Many existing indicators aim to assess status and trends in biodiversity and ecosystem integrity, but not the ecosystem services and benefits directly arising from these.

A clear and proven linkage between biodiversity, system integrity and ecosystem services is required to render indicators suitable for service status and trend assessment. In theory, indicators could also provide information on critical thresholds or tipping points, to function as any early warning system. However, such ecosystem service indicators would need to be ecosystem-specific to take account of what are likely to be different thresholds or tipping points in response to any given pressure. For most ecosystems, these ecological thresholds are unknown (Section 2.7).

The EU RUBICODE ("Rationalising Biodiversity and Conservation in Dynamic Ecosystems") project suggested criteria for assessing the suitability of existing indicators,[398] including whether:

- the purpose of the indicator has been defined;
- the indicator type is appropriate;
- the indicator is linked to biodiversity and ecosystem services;
- the spatial scale fits the purpose;
- a baseline or reference benchmark is definable; and,
- data/sampling protocols are available.

Biotic and Abiotic Indicators

Current indicators and targets have not necessarily been developed with an ecosystem service delivery purpose in mind. Most existing relevant indicators assess trends in biodiversity and habitat quality for the conservation strategies of particular species and habitats, the protection of human health, or have a particular policy sector focus (e.g. agriculture, forestry).

Biotic indicators, those referring to biological organisms, are required to assess status and trends in biodiversity. Abiotic measures, those referring to physical and chemical inputs (such as levels of a pollutant) are useful for detecting and quantifying levels of environmental stress impacting ecosystems. Biotic indicators are usually collected at habitat scales rather than regionally or nationally, and there are difficulties in scaling them up to the same scale at which some abiotic measures are made, such as earth observation satellite data.[403]

Monitoring of the status of single species, or groups of similar species, can also be a misleading indicator of ecosystem condition, as species groups will have differing sensitivity to degradation of ecosystems. Recent studies have suggested that proxies may be suitable for identifying broadscale trends in ecosystem services, but are unlikely to be suitable for identifying ecosystem service hotspots or priority areas for multiple ecosystem services.[404] Most existing indicators for large-scale biodiversity assessment use surrogate measures, such as land use change, but the applicability of surrogates to account for biodiversity and its different components is questionable if not linked to actual validated biodiversity levels.[403]

Remote satellite sensing data can identify the dominant species that create and maintain large area physical structures over long periods, such as forests, wetlands or coral reefs, from which assessments of biodiversity populations can be estimated. However, the extrapolation of these data to provide an indication of ecosystem service provision needs to be validated further, particularly to define the relationship between landscape metrics suggested for ecosystem accounts and measures of biodiversity components.[398]

Adapting Existing Suites of Indicators
A recent review of the available suite of indicators commissioned by Defra concluded that they do provide a good overview of the state of ecosystems. The study found gaps in terms of their ability to monitor the delivery of ecosystem services and, in particular, pressures on specific ecosystems. Research is required to develop indicators that cover the functional, structural and genetic components of biodiversity and their relationship to ecosystem service provision. [405]

The Defra review suggested that a framework for summary indicators could be constructed, based on: indicators for the state of different ecosystem types as a proxy for their overall functioning; indicators for ecosystem services (impacts); indicators of the pressures acting on ecosystems; and, indicators of the interactions between civil society, the environment and the economy (driving forces). This framework would consist of approximately 150 indicators, although further definition of indicators for particular ecosystem services is required.[405]

Figure 27: The Draft Classification of Ecosystem Goods and Services. [406]

SEEA 2003function	CICES Theme	CICES Class	TEEB Categories			
resource	Provisioning	**Food & Beverages**	Food	Water		
resource		**Materials**	Raw Materials	Genetic resources	Medicinal resources	Ornamental resources
resource		**Energy**				
sink	Regulating and Maintenance	**Regulation of waste assimilation processes**	Air purification	Waste treatment (esp. water purification)		
service		**Regulation against hazards**	Disturbance prevention or moderation	Regulation of water flows	Erosion prevention	
service		**Regulation of biophysical conditions**	Climate regulation (incl. C-sequestration)	Maintaining soil fertility		
service		**Regulation of biotic environment**	Gene pool protection	Lifecycle maintenance	Pollination	Biological control
service	Cultural	**Symbolic**	Information for cognitive development			
service		**Intellectual and Experiential**	Aesthetic information	Inspiration for culture, art and design	Spiritual experience	Recreation & tourism

Ecosystem service delivery indicators would allow the flow of benefits provided by natural capital stocks to be assessed as well as making explicit the impacts arising from changes in the capacity of ecosystems to provide these benefits, with subsequent effects on human well-being. However, there is a lack of indicators relating to the links between biodiversity and regulating and supporting ecosystem services.[407]

Ecosystem Service Indicators are also key to informing ecosystem management decisions (Chapter 4), but these would need to be more detailed, as well as at relevant spatial and temporal scales. The World Resources Institute is currently developing frameworks for the use of indicators and metrics that describe the status and trends for biodiversity and ecosystem service provision.[408] The Ecosystem Service Indicators Database is being developed to support policy decisions containing indicators which have been successfully used in ecosystem approaches to management.[409]

To derive a final set of indicators, ecosystem service benefits at the UK, country and regional scales will first need to be defined. Indicators are required that directly refer to components of biodiversity or the ecological processes underpinning ecosystem services. Most ecosystem services have few or no suitable indicators to monitor the actual delivery of services, although indicators of ecosystem function (Section 2.7) can act as indicators of potential service delivery.[405]

The Countryside Survey

The Countryside Survey provides a unique time series of systematically collected data which incorporate measures of soil, water, vegetation and landscape taken at the same locations. An integrated assessment of Countryside Survey (CSS) data to investigate Ecosystem Services in Great Britain identified 38 variables measured in the survey as potential indicators of ecosystem service provision, relating to different aspects of landscapes and ecosystems. This includes 11 for headwaters, 3 for ponds, 8 for soils, 10 for species diversity and 6 for cultural aspects of landscapes.[410]

The different indicators for any one ecosystem service developed from the CSS data were not always in agreement, suggesting a 'bundle' of biological and physical measurements may be required fully to assess ecosystem service categories. Overall trends could be identified for indicators classified by ecosystem compartment (soils, vegetation, waters, habitat extent and landscapes) and by ecosystem service category.[410] In general, the coverage of CSS indicators varied greatly between services but suggested regulating and supporting services were largely stable or improving, reflecting trends for waters and soils. However, the regulating service of pollination showed declines, as did indicators relating to cultural services, including plant diversity and landscapes. [410]

The CSS analysis of possible indicators highlighted the requirement for trade-offs to be made by local management solutions to optimise required levels of benefit delivery. For example, ecological succession (Section 5.1) from open to wooded habitats reduced plant diversity, affecting pollination services, but improved the biological quality of stream headwaters. However, analysis of the relationships between ecosystem service indicators suggests that local strategies to maximise delivery of 'bundles' of ecosystem service benefits are constrained by landscape scale trade-offs between productivity and species richness that reflect over-arching soil and climate gradients. The optimal situation for maximising mixed service delivery at the local scale is in areas with a high diversity of species and habitats, and where overall productivity is intermediate. [410]

Other key points of interest arising from the analysis included: [410]

- Biodiversity plays a role in delivery of ecosystem services and possible means of measuring and quantifying those services as well as resources required to maintain them and likely impacts of change.
- The ecosystem approach and integrated assessment can be used to help recognise interactions between ecosystems, delivery of services and the impacts of cross cutting policies.
- Interdisciplinary approaches are required to develop an evidence base to enable evaluations of possible policy options for management of natural resource systems and maintenance of ecosystem service benefits.
- Decisions and trade-offs to inform land management and use could be facilitated by developing models to predict possible future impacts of policy interventions (section 5.10).
- There are ecosystem service surpluses and deficits in different parts of the country. There is a need to increase understanding of the relationships with differences in human population, service demand and delivery in different parts of the country and expected changes in these in the future.
- There is a need to increase understanding of long term effects of drivers of change in

ecosystems, particularly time lags in responses to changes.
- Economic valuation could increase appreciation of, and improve management of, natural capital.
- Development and application of ecosystem service tools is not necessarily in conflict with previous approaches seeking to conserve biodiversity and charismatic landscapes for their intrinsic value, if cultural, spiritual and aesthetic services are taken into account.

Indicator Knowledge Gaps
Where studies have indicated there are particular pressures impacting a habitat type, such as land drainage and wet grassland, there are currently no available indicators to monitor those pressures. Many pressures on habitats result from land use change and, in terms of ecosystem service benefits, it is important that indicators not only reflect changes in habitat stock, but also what habitats are changing from and to. For example, agri-environment schemes (Section 5.6) could deliver an expansion in woodland habitat and the ecosystem service benefits arising from this, but if this woodland is created on other semi-natural habitats, such as heathland or moorland, it would have an impact on the delivery of other ecosystem service benefits.

The EU RUBICODE project suggested that there is a range of knowledge gaps that need to be addressed before a suitable set of indicators for monitoring ecosystem service provision could be defined, [398] including how to:
- measure structural and functional components of diversity in all ecosystems at relevant spatial scales;
- set comparable reference thresholds/quality targets for components of biodiversity;
- identify and measure key ecosystem function and processes;
- identify the linkage of these ecosystem functions/processes to all the categories of ecosystem service provision;
- identify critical service provision levels for sustaining human well-being;
- assess the status and trends of biodiversity and ecosystem services in all ecosystems; and,
- develop broadly applicable and integrated metrics for multiple ecosystem service provision that can be understood by policymakers, decision makers and the public.

Annex C Convention on Biological Diversity Principles of the Ecosystem Approach[411]

Principle 1: The objectives of management of land, water and living resources are a matter of societal choices.	Different sectors of society view ecosystems in terms of their own economic, cultural and society needs. Indigenous peoples and other local communities living on the land are important stakeholders and their rights and interests should be recognized. Both cultural and biological diversity are central components of the ecosystem approach, and management should take this into account. Societal choices should be expressed as clearly as possible. Ecosystems should be managed for their intrinsic values and for the tangible or intangible benefits for humans, in a fair and equitable way.
Principle 2: Management should be decentralized to the lowest appropriate level.	Decentralized systems may lead to greater efficiency, effectiveness and equity. Management should involve all stakeholders and balance local interests with the wider public interest. The closer management is to the ecosystem, the greater the responsibility, ownership, accountability, participation, and use of local knowledge.
Principle 3: Ecosystem managers should consider the effects (actual or potential) of their activities on adjacent and other ecosystems.	Management interventions in ecosystems often have unknown or unpredictable effects on other ecosystems; therefore, possible impacts need careful consideration and analysis. This may require new arrangements or ways of organization for institutions involved in decision-making to make, if necessary, appropriate compromises.
Principle 4: Recognizing potential gains from management, there is usually a need to understand and manage the ecosystem in an economic context. Any such ecosystem-management programme should: ■ **Reduce those market distortions that adversely affect biological diversity;** ■ **Align incentives to promote biodiversity conservation and sustainable use;** ■ **Internalize costs and benefits in the given ecosystem to the extent feasible.**	The greatest threat to biological diversity lies in its replacement by alternative systems of land use. This often arises through market distortions, which undervalue natural systems and populations and provide perverse incentives and subsidies to favour the conversion of land to less diverse systems. Often those who benefit from conservation do not pay the costs associated with conservation and, similarly, those who generate environmental costs (e.g. pollution) escape responsibility. Alignment of incentives allows those who control the resource to benefit and ensures that those who generate environmental costs will pay.
Principle 5: Conservation of ecosystem structure and functioning, in order to maintain ecosystem services, should be a priority target of the ecosystem approach.	Ecosystem functioning and resilience depends on a dynamic relationship within species, among species and between species and their abiotic environment, as well as the physical and chemical interactions within the environment. The conservation and, where appropriate, restoration of these interactions and processes is of greater significance for the long-term maintenance of biological diversity than simply protection of species.
Principle 6: Ecosystem must be managed within the limits of their functioning.	In considering the likelihood or ease of attaining the management objectives, attention should be given to the environmental conditions that limit natural productivity, ecosystem structure, functioning and diversity. The limits to ecosystem functioning may be affected to different degrees by temporary, unpredictable of artificially maintained conditions and, accordingly, management should be appropriately cautious.
Principle 7: The ecosystem approach should be undertaken at the appropriate spatial and temporal scales.	The approach should be bounded by spatial and temporal scales that are appropriate to the objectives. Boundaries for management will be defined operationally by users, managers, scientists and indigenous and local peoples. Connectivity between areas should be promoted where necessary. The ecosystem approach is based upon the hierarchical nature of biological diversity characterized by the interaction and integration of genes, species and ecosystems.
Principle 8: Recognizing the varying temporal scales and lag-effects that characterize ecosystem processes, objectives for ecosystem management should be set for the long term.	Ecosystem processes are characterized by varying temporal scales and lag-effects. This inherently conflicts with the tendency of humans to favour short-term gains and immediate benefits over future ones.
Principle 9: Management must recognize	Ecosystems change, including species composition and population abundance. Hence,

the change is inevitable.	management should adapt to the changes. Apart from their inherent dynamics of change, ecosystems are beset by a complex of uncertainties and potential "surprises" in the human, biological and environmental realms. Traditional disturbance regimes may be important for ecosystem structure and functioning, and may need to be maintained or restored. The ecosystem approach must utilize adaptive management in order to anticipate and cater for such changes and events and should be cautious in making any decision that may foreclose options, but, at the same time, consider mitigating actions to cope with long-term changes such as climate change.
Principle 10: The ecosystem approach should seek the appropriate balance between, and integration of, conservation and use of biological diversity.	Biological diversity is critical both for its intrinsic value and because of the key role it plays in providing the ecosystem and other services upon which we all ultimately depend. There has been a tendency in the past to manage components of biological diversity either as protected or non-protected. There is a need for a shift to more flexible situations, where conservation and use are seen in context and the full range of measures is applied in a continuum from strictly protected to human-made ecosystems
Principle 11: The ecosystem approach should consider all forms of relevant information, including scientific and indigenous and local knowledge, innovations and practices.	Information from all sources is critical to arriving at effective ecosystem management strategies. A much better knowledge of ecosystem functions and the impact of human use is desirable. All relevant information from any concerned area should be shared with all stakeholders and actors, taking into account, inter alia, any decision to be taken under Article 8(j) of the Convention on Biological Diversity. Assumptions behind proposed management decisions should be made explicit and checked against available knowledge and views of stakeholders.
Principle 12: The ecosystem approach should involve all relevant sectors of society and scientific disciplines.	Most problems of biological-diversity management are complex, with many interactions, side-effects and implications, and therefore should involve the necessary expertise and stakeholders at the local, national, regional and international level, as appropriate.

Annex D: Impact Assessments and Planning Decisions

D1 Regulatory Impact Assessment

In the UK, any department or agency in government wanting to enact a new policy measure of a regulatory nature likely to have a significant economic, environmental or social impact needs to complete a detailed Impact Assessment (IA).[412] The IA guidance requires that the government explains exactly why a proposed intervention is necessary and why it was selected over alternatives. The checklist of Specific Impacts Tests is available on the Department for Business, Innovation and Skills (BIS) website.[413] IAs apply to primary and secondary legislation, and legislation to enact EU policy as well as to codes of practice or guidance. Spending proposals require an IA only where they involve an administrative or regulatory burden. The minister responsible for the policy (or chief executive of an agency or NDPB) is required to sign off public Impact Assessments.

As a first step to policy frameworks for assessing environmental impacts, a completely revised 'Sustainable Development Specific Impact Test' and a 'Wider Environment Specific Impact Test' have been developed for use in Impact Assessments. A policy, or the cumulative impact of many policies in combination with other outside factors, could lead to environmental limits being impacted, and the Sustainable Development test includes a requirement to consider the policy's impact on statutory environmental limits.

The tests include a combination of monetised and non-monetised costs and benefits, with valuations of environmental changes taking account of environmental limits as far as possible. To ensure acceptable levels of ecosystem services are maintained, a checklist of ecosystem services is incorporated into the Wider Environmental Specific Impact Test.

These tests apply only to policies requiring an Impact Assessment, although they are expected to be more widely used once formalised in updated Green Book guidance.[414] Cost-benefit analysis (CBA) is a focal element in a range of the Specific Impact Tests used in the assessment process, which also assesses the associated risks that a proposal might have an impact on the public, private or third sector, using the Green Book's appraisal and evaluation techniques.

Once a policy is implemented it should continue to be regularly assessed to inform further policy development in line with the broad stages of the policy cycle known as ROAMEF (Rationale, Objectives, Appraisal, Monitoring, Evaluation and Feedback), as highlighted in the Treasury's Green Book. The assessment is formally published only at certain points in the policy cycle. The IA must be first published when the government decides on policy option (usually after a consultation exercise) and when a government, or private member's bill enjoying government support, is introduced in either House of Parliament. A further IA should be carried out if changes have been introduced to legislation during the parliamentary process. An IA should also be published when a draft statutory instrument is laid in Parliament.

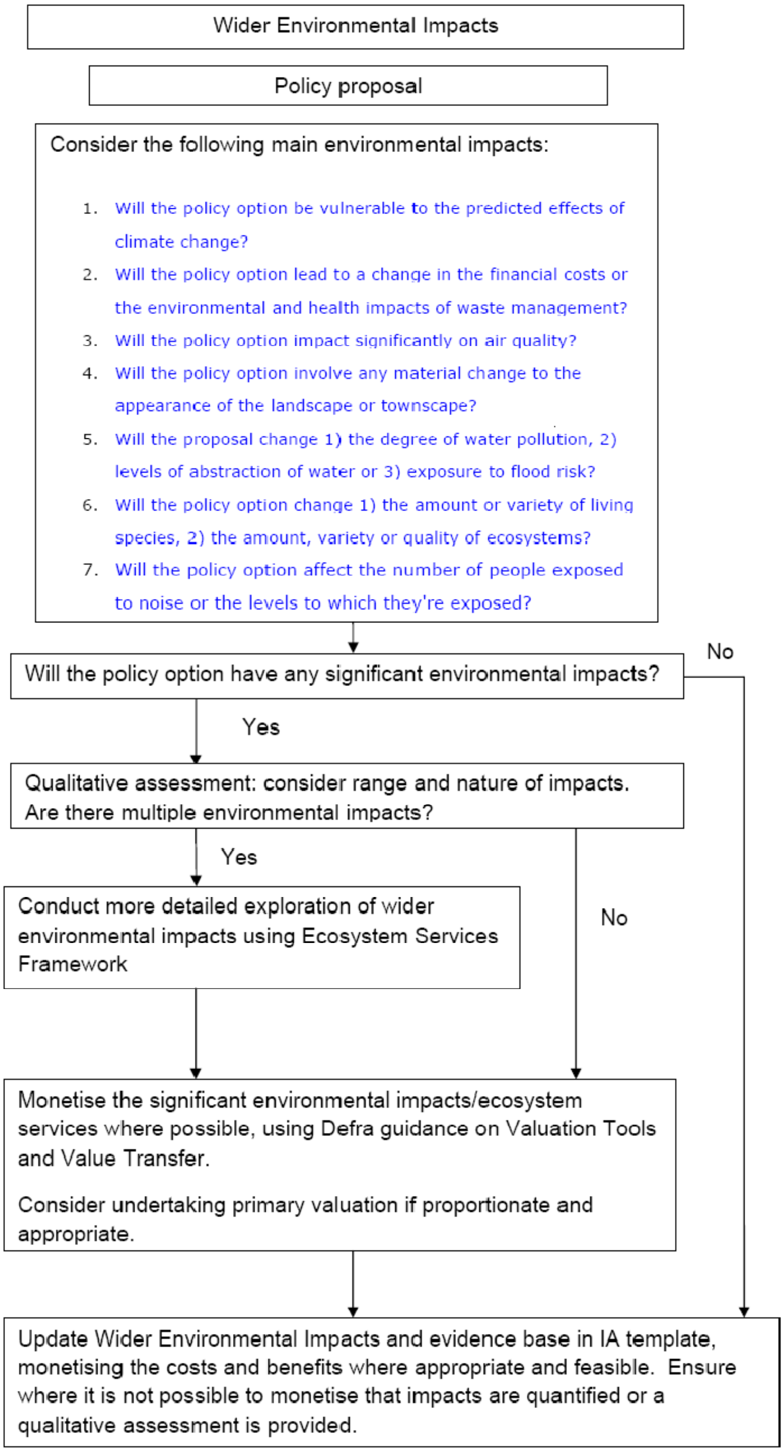
In these ways, IAs are subject to parliamentary scrutiny and debate. These tests will include assessments of impacts on levels of ecosystem service benefit (Annex D). Although decisions on land use may be taken in line with policies, legal obligations and other factors, the Green Book states that decisions on policy options resulting in land use change should reflect the balance of costs and benefits associated with different possible land uses and changes in use, and it is this balance that determines the value of land in different uses. The guidance also recognises that lack of relevant information on values may result in uncertainty and that the results of sensitivity and scenario analyses should also generally be included in presentations and summary reports to decision-makers, rather than just single-point estimates of expected values.

D2 Wider Environmental Specific Impact Test

The impact of policy options on the environment should be considered from the outset of the policy appraisal process, with a preliminary assessment of the environmental impacts using the wider environmental issues checklist of questions (Figure 28). If significant environmental impacts are detected, further work is required to identify the importance of the overall policy decision, to quantify and value the environmental impacts and to highlight key uncertainties and evidence gaps that require further investigation.

Assessment should take sufficiently broad account of the possible spatial scale and temporal nature of the impacts, which may differ from those of the initial policy decision. After a qualitative assessment of the impacts, which includes a description of changes in quantity or quality; whether the change will contribute to a deterioration or improvement in the environment; the temporal nature of the change (for example limited, permanent or gradual), the spatial location; and, the

Figure 28 Wider Environmental Specific Impact Test Flow Diagram



scale of the change, a more detail quantitative assessment may be necessary.

In cases where there are multiple environmental impacts, affecting both market and non-market values, the use of the ecosystem services framework is recommended (Section 4.5). Policy options should be assessed against the Defra checklist of ecosystem services (Figure 29) to ensure that all ecosystem services are included from the outset and the whole ecosystem is considered. By identifying the impact on ecosystem services, it is possible to identify the impacts on human welfare of these changes.

Where significant environmental impacts of a policy option have been identified, the impacts should be

Figure 29: The Defra Checklist of Ecosystem Services for Consideration in an Ecosystem Service Valuation Assessment (ESVA).

Category	Baseline/ 'Do nothing' policy option 0	Policy option 1	Policy option 2	Policy option 3
Provisioning services				
Food				
Fibre and Fuel				
Genetic resources				
Biochemicals, natural medicines, pharmaceuticals				
Ornamental resources				
Fresh water				
Regulating services				
Air-quality regulation				
Climate regulation				
Water regulation				
Natural hazard regulation				
Pest regulation				
Disease regulation				
Erosion regulation				
Water purification and waste treatment				
Pollination				
Cultural services				
Cultural heritage				
Recreation & tourism				
Aesthetic value				
Supporting Services				
Soil formation				
Primary production				
Nutrient cycling				
Water cycling				
Photosynthesis				

Score	Assessment of effect
++	Potential significant positive effect
+	Potential positive effect
0	Negligible effect
-	Potential negative effect
--	Potential significant negative effect
?	Gaps in evidence

monetised where possible. As a first step, they should be quantified in terms of magnitude and over time by direct or proxy measures on the basis of scientific and technical evidence. Monetary valuation can be informed by use of the Total Economic Valuation framework to understand whether values are use values or non-use values and what non-market valuation approaches are appropriate (Section 4.7).

It is unlikely that all the economic values associated with changes in environmental impacts can be estimated, but quantitative and qualitative information can be used to inform evidence gaps. As far as possible, impacts should be quantified and valued and this information presented in the main policy appraisal so that environmental costs and benefits can be compared with all other costs and benefits of a policy option.

Sustainable Development Specific Impact Test

The role of the Sustainable Development Specific Impact Test is to enable government departments conducting policy appraisal to identify key impacts of their policy options relevant to sustainable development and to give informed advice to ministers on sustainability-related issues. Because sustainable development cannot be adequately appraised by cost benefit analysis alone, this test provides a framework within which to combine information about sustainable development impacts with information from the rest of the impact assessment about the balance of monetised and non-monetised costs and benefits. The findings of the wider environmental impacts test should be used to inform the environmental standards section of the test, which also includes consideration of intergenerational impacts and how discounting should be applied (Figure 30). Social impacts will be included in forthcoming versions of the test.[415]

The test includes a specific requirement to consider whether a policy proposal may contribute to breaching environmental standards (section 2.2). If so, the the government department which has legal responsibility for the threshold should be notified and advice sought on:

- how to account for the impact;
- whether to inform ministers of the risk; and,
- how to identify mitigating or compensating actions where appropriate.

If the IA test suggests that the balance of monetised and non-monetised costs and benefits has a net positive impact then the policy should proceed, unless there are issues relating to statutory environmental standards or long term impacts. In the case of issues around environmental standards, there should be consideration of whether any necessary changes or additions to the policy affect the conclusion of the IA. Where there are significant long-term impacts, these should be presented to ministers clearly, alongside any mitigating actions where appropriate, to allow them to decide whether or not the strength of the net benefits implied by the IA result is sufficient to outweigh any negative long-term impacts.[415]

There will be several infrastructure decisions to be taken by government in the next five years that have significant sustainable development implications and that will have impacts on UK natural resource systems. The government published the National Infrastructure Plan 2010 on 25 October 2010. This set out a number of key priorities and plans for UK infrastructure, including for energy, transport, digital communications, flood management, water and waste. There was also a commitment to publish a more detailed version of the plan by the end of 2011, setting out the long-term investment needs and priorities for economic infrastructure for the UK, along with the priority actions to deliver them, including:

- establishing a common set of planning assumptions, such as economic growth forecasts, population growth forecasts, impacts of climate change
- identifying relevant constraints, including establishing a framework for assessing overall affordability.

The Treasury will also publish high level supplementary Green Book guidance on assessing the costs and benefits of policy and spending interventions related to economic infrastructure in early 2011, building on the its Five Case Model for business cases and existing Green Book guidance. This will set out how to:

- Assess investment need, funding models and financing of economic infrastructure at a strategic level (i.e. as programmes rather than as specific projects).
- Identify and evaluate constraints on the ability to deliver projected investment programmes (including exchequer and distributional impacts, affordability, access to finance, supply chain capacity, environmental impact and resilience).
- Assess the most appropriate responses to address these constraints to ensure the right investment is prioritised at the right time using Green Book methodology.
- Assess and value significant downside risks associated with infrastructure decisions, e.g. security of supply issues created by delaying or deferring investment.
- Deal with the uncertainty in quantifying long-term costs and benefits of major infrastructure programmes, especially where impacts are transformational.

D3 Planning Policy and Environmental Impacts

A detailed analysis of how the present land use planning system operates for different sectors of land use, at different spatial scales and at different levels of governance and the range of ‘systemic’ issues that need to be addressed to cope with future environmental and social issues were set out in the recent Foresight Report on “Land Use Futures”.[121] The recent “Making Space for Nature: A Review of England’s Wildlife Sites” and Ecological Network’ report also recommended that:[57]

- Local authorities should ensure that ecological networks, including areas for restoration, are identified and protected through local planning.
- Government should support local authorities in this role by clarifying that their biodiversity duty includes planning coherent and resilient ecological networks.
- Planning policy and practice should provide greater protection to other priority habitats and features that form part of ecological networks, particularly local wildlife sites, ancient woodland and other priority Biodiversity Action Plan habitats.

  Ecological Restoration Zones that operate over large, discrete areas within which significant

Figure 30 Defra Sustainable Development Impact Test

Stage 1

1. Environmental Standards

1a. Are there are any significant environmental impacts of your policy proposal (see Wider Environment Specific Impact Test)?
Yes No
If the answer is 'yes' make a brief note of the impacts below:

1b. If you answered 'yes' to 1a., are the significant environmental impacts relevant to any of the legal and regulatory standards identified?
Yes No
If the answer is 'yes' make a brief note of the relevant standards below:

If you answered 'yes' to 1b, have you:
1c. Notified the Government Department which has legal responsibility for the threshold and confirmed with them how to include the impacts appropriately in the analysis of costs and benefits?
1d. Informed ministers where necessary?
1e. Agreed mitigating or compensatory actions where appropriate?

2. Intergenerational impacts

2a. Have you assessed the distribution over time of the key monetised and non-monetised costs and benefits of your proposal? This assessment can be included in your Evidence Base or put in an annex.	
Yes	No

2b. Have you identified any significant impacts which may disproportionately fall on future generations? If so, describe them briefly.	
Yes	No

If you answered 'yes' to 2b. , have you:
2c. Informed ministers where necessary? If so, provide details.
2d. Agreed mitigating or compensatory actions where appropriate? Provide details.

Stage 2

3. The purpose of the second stage is to bring together the results from the impact assessment with those from the first stage of the SD test. The following questions are intended to reflect the uncertainties in the cost benefit analysis and help you consider how to proceed in the light of further evidence from the first stage of the SD test.

3a. Indicate in the appropriate box whether the balance of monetised costs and benefits is:				
Strongly positive	Moderately positive	Roughly neutral / finely balanced	Moderately negative	Strongly negative

3b. Indicate in the appropriate box whether the balance of non-monetised costs and benefits is likely to be:				
Strongly positive	Moderately positive	Roughly neutral / finely balanced	Moderately negative	Strongly negative

3c. Indicate in the appropriate box whether the results of the SD questions 1-3 are, on balance, likely to be:				
Strongly positive	Moderately positive	Roughly neutral / finely balanced	Moderately negative	Strongly negative

3d. Indicate in the appropriate box whether, overall, the balance of the monetised and non-monetised costs and benefits and the sustainability issues is considered to be:				
Strongly positive	Moderately positive	Roughly neutral / finely balanced	Moderately negative	Strongly negative

3e. Provide an explanation of the final result from 3d, explaining, for example, how you have compared monetised and non-monetised costs and benefits and how you have resolved any conflicts between the cost-benefit results and the SD results.

enhancements of ecological networks are achieved, by enhancing existing wildlife sites, improving ecological connections and restoring ecological processes, should be implemented by consortia of local authorities, local communities and landowners, the private sector and voluntary conservation organisations, supported by national agencies.
- The government should ensure that the remaining areas of high conservation value that currently are not well protected are effectively safeguarded.
- When determining the boundaries of designated sites, responsible authorities should take better account of the need to support underpinning ecological processes and of anticipated environmental change.
- Public bodies and other authorities responsible for canals, roads, railways, cycle ways and other linear features should ensure that they achieve their potential to be wildlife corridors, thereby enhancing the connectivity of ecological networks.

There are approximately half a million planning applications a year, about 15,000 of which are classed as major developments. Between June 2009 and June 2010, there were 311 planning applications that fell within the legislative requirements for an Environmental Impact Assessment (EIA). Increases in population and the number of households are likely to continue to raise demand for land use change in the UK. Higher household disposable incomes, car ownership, and home-to-work commuting will also influence demand for land for transport and other infrastructure, such as shopping complexes. There are also likely to be new demands for land, such as land for new energy infrastructure, energy crops, or to adapt to climate change, such as coastal retreat.

England has traditionally regulated market demand for land use through the planning system, although this applies only to a subset of land use change, development and infrastructure. The Planning Act 2008 introduced a new set of national policy guidance for specified descriptions of nationally-significant development, National Policy Statements (NPS) to be produced by the relevant government departments. All the NPSs are subject to parliamentary scrutiny through the relevant select committee that scrutinises the government department producing the NPS.

The NPSs were originally intended to integrate the government's objectives for infrastructure capacity and development with its wider goal to deliver the UK's five sustainable development principles in England, including environmental limits. The Spatial Planning System in England operated at three statutory tiers, NPSs, Regional Spatial Strategies and Local Development Frameworks. The NPSs were initially to inform regional spatial strategies, established by the Planning and Compulsory Purchase Act 2004 to provide a broad development strategy for a region for a fifteen to twenty year period, including identifying priorities for the environment. They set out the spatial strategy for the sustainable development of the region, with the plans subject to Strategic Environmental Assessment, Habitat Regulations Appropriate Assessment and Sustainability Appraisal (Section D4, Box 25). The present government revoked regional spatial strategies in 2010, but will require primary legislation to abolish them following a High Court ruling in 2011.

As well as abolishing Regional Spatial Strategies, the Decentralisation and Localism Bill contains reforms to planning to give greater powers to local communities and members of the public over planning decisions.[416] There is an intention that planning is only used as a mechanism for delivering Government objectives 'where it is relevant, proportionate and effective to do so'. Nationally significant infrastructure will be overseen by the major infrastructure planning unit in the Planning Inspectorate, replacing the Infrastructure Planning Commission, with final decisions now taken by the Secretary of State. Northern Ireland, Wales and Scotland retain different planning frameworks, with Wales retaining spatial planning and Scotland introducing a land use strategy to the Scottish Parliament in March 2011.[417]

Box 25 Sustainability Appraisal

The mechanism for carrying the required SEA assessment of Local Development Frameworks and Regional Spatial Plans was integrated into the 'Sustainability Appraisal' (SA) of the draft plans in England and Local Development Plans in Wales. The SA included a wider range of considerations than the environmental impact focus of the SEA, including the economic and social impact of plans. While the SA was concerned with promoting an integrated approach to sustainable development, covering social, environmental and economic issues, it did not reflect the loss of critical natural capital. The SA did report on whether planned developments will hinder or enhance progress towards environmental objectives, but did not consider whether the cumulative impacts of development will contribute to the future breaching of environmental thresholds.[418] The criteria for judging significance of cumulative effects are not different from those for other types of environmental assessment, but threshold effects and irreversible changes in the use of critical natural capital will generally be key concerns. One well known example of cumulative impacts analysis is that carried for the Stern report on the cumulative impacts of climate change

Habitat Regulations Appropriate Assessment (HRA)

An assessment is required where a plan (including land use plans at the national or regional level) or project may have a significant effect on a site protected under the EU Habitats Directive. As the SEA Directive encompasses plans and programmes likely to have significant effects on sites designated the Habitats Directive, Habitat Regulation Appropriate Assessment (HRA) can be carried out alongside SEAs where applicable. If there is likely to be an adverse impact on the integrity of the designated site, the plan or project cannot be approved, unless no alternatives and overwhelming social and economic requirements, in which case compensatory habitat can be created or restored (Section 5.3).

Planning Policy Statement 9

At the local and regional level, Section 39 of the Planning and Compulsory Purchase Act 2004 placed a statutory duty on all persons or bodies exercising functions with respect to regional spatial strategies and local development plans to contribute to further the UK Sustainable Development Strategy. These bodies included the regional planning bodies and local planning authorities and they must have regard to Planning Policy Statement 1: Delivery of Sustainable Development and Planning Policy Statement 9 (PPS9): Biodiversity and Geological Conservation. PPS9 requires that planning authorities:

- have up-to-date information about the environmental characteristics of their area;
- include form and location information about developments in the planning process to take a strategic approach to biodiversity;
- provide opportunities for incorporating and enhancing biodiversity in development proposals;
- take planning decisions that prevent harm to biodiversity and geological features; and,
- should refuse planning permission if significant harm to "biodiversity interests" cannot be prevented, adequately mitigated against, or compensated for.

Biodiversity "interests" were not clearly defined in PPS9, but are understood to include sites with statutory conservation designations, UK Biodiversity Action Plan species and habitats, locally designated sites, ancient woodlands and veteran trees and "important habitat networks". The accompanying guidance also suggested that planning authorities should ensure that planning applications are submitted with adequate information using early negotiation, published checklists, and requiring ecological surveys and appropriate consultation.

However, it should be noted that the wording of PPS9 is sufficiently loose that economic and social considerations often override ecological considerations when a site of ecological importance is developed. The government intend to consolidate all existing planning policy statements, circulars and guidance documents into a single "National Planning Policy Framework" (NPPF). The framework will also contain a presumption in favour of sustainable development.[416] Organisations such as the Woodland Trust have raised concerns that the loss of PPS9 may result in significant loss of habitats and biodiversity.

There is also a "Biodiversity Duty' placed on planning authorities, under section 40 of the Natural Environment and Rural Communities (NERC) Act, 2006, which requires that: "every public authority must, in exercising its functions, have regard, so far as is consistent with the proper exercise of those functions, to the purpose of conserving biodiversity". A recent review of this duty concluded that, although the duty had generally had a positive impact, performance and understanding are very variable across local authorities. The review recommended that the government should take action to support improved implementation.[419] The recent "Lawton" review recommended the government should clarify the duty to include planning for coherent and resilient ecological networks.[57]

Green Infrastructure

It is notable that some Regional Spatial Strategies contained consideration of natural resources, often referred to as 'green' infrastructure, although there was no consistent approach to natural resource systems and related ecosystem services. A green infrastructure approach differs from conventional open space planning, because it seeks to consider multiple functions and benefits of landscapes in concert with land development, growth management and built infrastructure planning.

The North West of England Regional Spatial Strategy, defined green infrastructure as " …the network of natural environmental components and green and blue spaces that lies within and between the Northwest's cities, towns and villages and which provides multiple social, economic and environmental benefits".[420] Initiatives, such as Green Northwest, supported by agencies such as Natural England and the Environment Agency, are seeking to promote green infrastructure as a critical infrastructure, alongside transport, waste, and energy and to join up built infrastructure with natural processes in complementary ways.[421]

The 2008 Planning Act also contained provisions to empower local councils to apply a Community Infrastructure Levy (CIL) on new developments in their areas to support infrastructure delivery, including green infrastructure within those developments. However, to fund wider local and regional green infrastructure requirements would require an additional funding mechanism. One option would be to extend section 106 agreements (which provide a mechanism to do this). However, significant justification is required for the application of Section 106 to developers, in addition to CIL. It is also not clear how regional green infrastructure requirements will be identified following abolition of regional spatial strategies.

Strategic Nature Areas

As a key part of its approach to 'green infrastructure', the South West Regional Spatial Strategy included a "nature map" which used available biodiversity monitoring datasets to identify blocks of land, referred to as 'Strategic Nature Areas' (SNAs) at the landscape scale (Figure 14). They may comprise a number of formally designated areas for conservation as well as land that has no designation for biodiversity conservation.

Traditional farming methods, together with climatic conditions and the underlying geology, have produced distinctive regional landscapes in the UK. The local landscape is shaped by natural landforms, local building materials, species and habitat types and land management practices. These have combined to create distinctive and unique character areas in the UK and are used as the basis for Landscape Characteristic Assessment, which identifies the features that give a locality it's "sense of place" and pinpoints what makes it different from neighbouring areas.

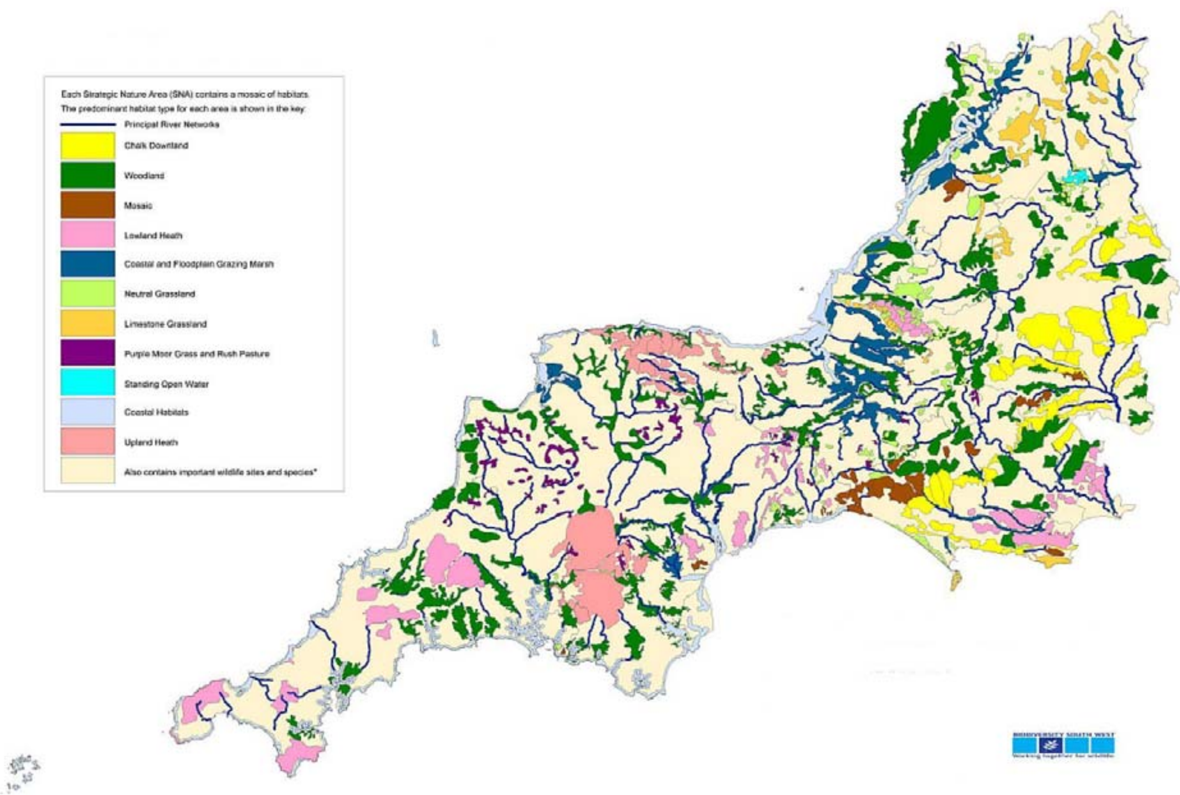
SNAs are a means of applying a Landscape Character Area approach at the landscape scale. The landscape character process analyses layers of ecological and cultural data including landform, the underlying geology and soils, patterns of settlement, land cover and differences in tree cover and is applied at the local scale through Landscape Description Units (LDUs), the building blocks of Landscape Character Areas. [422] The map was created by Biodiversity South West, tasked with delivering the UK Biodiversity Plan for the region.

SNAs represent the remaining fragments of what was once much more extensive areas of natural and semi-natural habitat. These fragments are often the most important features in the landscape for nature conservation and provide the best characterisation of an area's wild flora and fauna. The mapping approach aims to identify areas of broadly similar character, distinct from their neighbours, along with the characteristic patterns and features within them. The Strategic Nature Areas show the spatial distribution and pattern of priority habitats that are characteristic of the different sub-regions of the South West. This indicates which habitat, of regional importance, is most appropriate for restoration and expansion in any particular area.[422]

The main aims are that the map should:

- identify where most of the major biodiversity concentrations are found and where targets to maintain, restore and re-create wildlife might best be met;
- inform the formulation and use of appropriate policies by planners at the local and regional level
- assist in targeting agri-environment scheme (section 5.8).

Figure 31: The Strategic Nature Areas in South West England Prioritised for the Conservation, Creation and Connection of Wildlife Habitats.[422]



The regional spatial strategy specifically stated "…provision for the maintenance, restoration and enhancement of habitats and species should be a significant component in the provision of green infrastructure". It also referred to the ecosystem approach, stating that:

- "it is important that targets for maintenance, restoration and recreation of priority habitats are met, taking an ecosystem approach, including opportunities for linking and buffering habitats and making them more coherent units."
- "local authorities should use the development process positively to achieve these outcomes and should promote beneficial management of priority habitats and species found in their areas. This should result in more resilient habitat units across the region".

The Biodiversity South West provided guidance for local planning authorities on how data can best be used to build up the necessary evidence base that will enable development proposals that take account of Strategic Nature Areas to be prepared. [422] Although this initiative will come to an end if regional spatial strategies are abolished, it provides a clear example of how a strategic approach to appraising ecosystem service benefits at a regional level could be incorporated into the planning system. Similar initiatives are being pursued using the information held by local biodiversity record centres to identify priority areas for the restoration of habitats and for agri-environment expenditure in counties such as Hampshire.

D4 Strategic Environmental Assessment

The central purpose of a strategic environmental assessment (SEA) is to anticipate and reduce the overall environmental impacts of proposed patterns of spatial development, and of multiple individual projects. It may be applied to an entire sector (e.g. a national energy policy) or to a geographical area (e.g. a Regional Spatial Strategy). The SEA Directive 2001/42/EC applies to plans and programmes made by authorities or utilities acting as authorities, both onshore and offshore, and therefore SEA is required both for government plans and programmes and for those of regional and local authorities.

The SEA was introduced into UK law through regulations for England, Wales, Scotland and Northern Ireland. It is enacted in Wales and England through the Environmental Plans and Programmes Regulations (Welsh SI 2004/1656 and SI 2004/1633). In Scotland, the Environmental Assessment (Scotland) Act 2005 widens the range of strategic actions to which SEA applies beyond those in the Directive, unlike in the rest of the UK, where the regulations closely follow the requirements of the Directive.

The SEA consists of a set of procedures relating to the provision of information and public consultation (including the preparation of an environmental report). These procedures are intended to provide a systematic and comprehensive process of identifying and evaluating the environmental consequences of proposed policies, plans or programmes to ensure that they are fully included and addressed early on in decision-making, along with economic and social considerations.

An SEA can provide information on environmental impacts arising from policy options, but essentially it is one of the processes in the production of a plan, policy or programme, and the environmental report output is often assumed to be more informative than it actually is. However, in contrast to an environmental impact assessment (EIA), an SEA provides decision-makers with information and strategies on a large scales and takes an holistic approach that considers projected environmental impacts of multiple actions over time. It has been suggested that a more explicit consideration of the ecosystem approach within SEAs would support the development of more integrated EIAs. [423]

The SEA Directive includes a requirement to provide information about measures "envisaged" to prevent, reduce or offset significant effects on the environment, but no specific mention is made of ecosystem services or biodiversity. They are also a recognised tool for comparing alternative options and the degree to which these meet environmental protection objectives. SEA of policies, plans and programmes, at national and local government levels is a possible existing regulatory mechanism for taking account of environmental limits, but the lack of clearly defined environmental limits is a major impediment to their incorporation. Theoretically, environmental limits should be a key part of defining a environmental baseline for an SEA, but there needs to be a clearly agreed basis for defining these in any given relevant situation.

Severn Barrage SEA

There is no systematic consideration of environmental limits or the likely impacts of biodiversity loss on ecosystem service provision at present in SEAs, although these could be incorporated as methodologies evolve. SEAs require environmental impacts to be taken into account during the planning phase of programmes that, among other criteria, are likely to affect significantly a nature conservation site protected under the Habitats Directive (92/43/EEC). Where impacts are found to be unavoidable, mitigation and compensation measures will be considered. It has also been suggested that if the value of ecosystem goods and services could be expressed in robust ways, it could form an essential element of SEA.

For example, the Department of Energy and Climate Change conducted a Severn Tidal Barrage Feasibility Study Consultation to inform the government decision whether to support tidal power development in the Severn estuary, which forms the border between South West England and South Wales. An SEA has been carried out in support of the government's Severn Tidal Power Feasibility Study to describe the likely significant effects on the environment of five shortlisted of ten possible tidal power projects within the Severn Estuary (Cardiff to Weston Barrage, Shoots Barrage, Beachley Barrage, Welsh Grounds Lagoon and Bridgewater Bay Lagoon, Figure 32). In addition, an ESVA and ESV was carried out by Eftec on behalf of DECC as part of the Feasibility Study to show the impact of the projects on ecosystem services within the estuary.

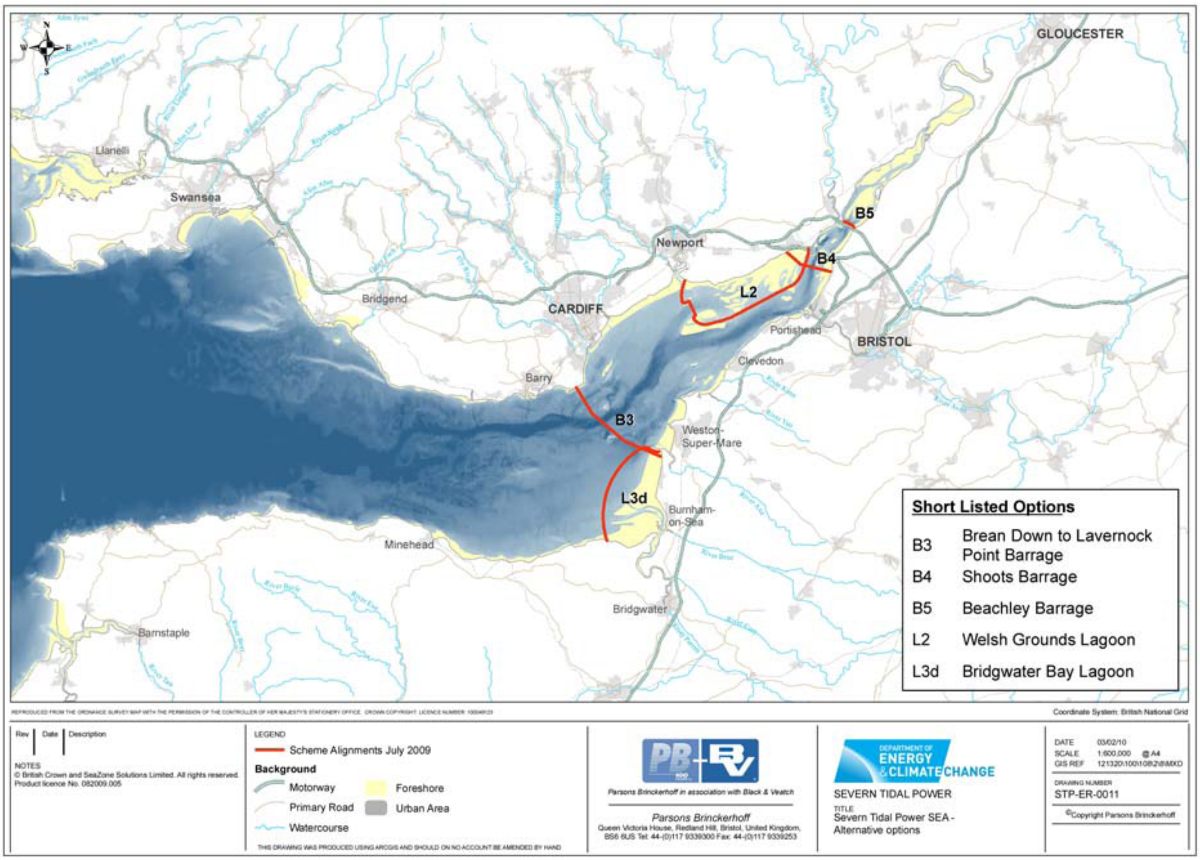
A tidal barrage in the Severn Estuary could provide nearly 5% of the UK's electricity and contribute to the country's long-term CO_2 emissions reductions targets. Any project to generate electricity from the tidal range of the Severn Estuary would need to meet the following objectives:[424]

- To generate electricity from the renewable tidal range resource of the Severn Estuary in ways that will have an acceptable overall impact on the environment and economy both locally and nationally, that will meet statutory obligations to nature conservation and provide benefit to the UK.
- To deliver a strategically significant supply of renewable electricity, which is affordable and represents value for money compared with other sources of supply, in the context of the UK commitments under the EU Renewable Energy Directive and Climate Change Act to deliver a secure supply of low carbon electricity.

The estuary is characterised by its 12.3 metre tidal range, the second largest in the world, and the movement of very large quantities of mud and sand between the seabed and in suspension in the water column. The Severn Estuary and the Bristol Channel provide one of the highest nutrient inputs into the marine environment in the UK. The unique environment of the Severn Estuary creates unusual physical conditions which influence the composition, distribution and quantity of its plants and animals.

The consequent biodiversity importance of the estuary has been recognised through the designation of a Special Protection Area (SPA) under the EU Birds Directive (09/147/EC) and it has been designated a Site of Community Interest by the European Union under EU Habitats Directive (92/43/EEC).

Figure 32: The Severn Estuary and the Location of the Proposed Barrage Schemes[424]



SPAs and SACs are assigned to areas of importance for national and international biodiversity, and together form the EU Natura 2000 network, designed to protect the most seriously threatened habitat and species. The Severn Estuary/Môr Hafren Special Area of Conservation supports mudflats and sandflats not covered by seawater at low tide, Atlantic salt meadows (salt marshes), sandbanks which are slightly covered by seawater all the time and *Sabellaria alveolata* reefs (sand reefs built by a worm species). The Severn Estuary, and the Rivers Tywi, Usk and Wye (all SACs) support seven migratory fish species protected under the EU Habitats Directive, notably allis and twaite shad, sea and river lamprey and Atlantic Salmon (the last of these not applying to the River Tywi).[424]

The Severn Estuary is designated an SPA, as it supports a number of qualifying populations of species of waterbird, including at least six waterbird species occurring in internationally important numbers (ringed plover, curlew, dunlin, pintail, redshank and shelduck), with the overall water bird assemblage using the Severn Estuary during winter calculated to be nearly 73,000 individual birds. In accordance with the Convention on Wetlands of International Importance (the Ramsar Convention), the Severn Estuary is classified as a Ramsar site due to its support of unusual estuarine communities, importance for all seven migratory fish species and as a habitat for wetland birds. In addition to the international and European designations, the estuary is recognised as a Site of Special Scientific Interest (SSSI) as it contains seagrass habitats and key bird breeding areas.[424]

Significant effects were determined using professional judgement taking account of the probability, duration, frequency and reversibility of the effects; their magnitude and spatial extent and the value and vulnerability of the area likely to be affected. These included:[424]

- The relationship between tidal range, sediment mobility and dynamic intertidal habitats. The tidal range could be reduced by up to 50% in some areas of the estuary by the Cardiff to Weston barrage project. It is likely that increased stand time of water upstream of a barrage, combined with the overall lower water level, will lead to enhanced erosion of the intertidal range and salt flats caused by wind generated wave action – as has been observed on the banks of the Dutch Eastern Schelde tidal barrage. This leads to the formation of saltmarsh cliffs that are no longer subject to the tidal inundation depositing wet sediment, leading to coarser or sandier bank habitats on the estuary. These enhanced erosion processes would lead to the deposition of fine sediment at the bottom of the estuary, reducing the amount of sediment and sand available for processes such as the formation of new mudflats further downstream. The extent of impact varies between alternative options, but significant negative effects on subtidal *Sabellaria alveolata* reefs as a result of reductions in flow speed are predicted to occur for any of the options except the Bridgewater Bay Lagoon.
- Over-wintering migratory birds that feed on the mudflats and salt marshes. Theseare likely to be affected negatively by changes to intertidal mud or saltmarsh habitat as a result of changes to water levels and sedimentation in all alternative options. Increased stand time of water within the estuary would reduce the available foraging time for birds in intertidal habitats, and the reduction of sea-water inundation upstream would increase freshwater influence and the displacement of existing saltmarsh vegetation by freshwater plant species, reducing intertidal habitat.
- Migratory fish species. These would all be affected by alterations to migratory cues and disruption of route of passage under all the alternatives. Fish may also be affected by habitat change and/or loss and injury from the tidal power schemes, varying between projects.
- Commercial fishing and recreational angling, with loss of fisheries under some options.
- The effects of the barrage on the Severn Bore and flood risk, particularly downstream of the barrage. Some 90,000 properties and commercial assets are at risk of flooding in over 500km^2 of low lying tidal floodplains of the Severn Estuary, concentrated in the urban centres of Cardiff, Newport, Burnham-on-Sea and Weston-super-Mare.
- The impact on marine water quality. The deposition of large quantities of sediment within the deeper, less active parts of the impounded area would result in increased light penetration and subsequent increase in plant growth and decay, affecting oxygen levels and water quality.

Overall, the ecosystem would shift from one that is heterogenous and highly dynamic to one that is more homeogenous, with a consequent reduction in levels of biodiversity and the ecosystem services arising from the dynamic processes occurring within the estuary. However, it is likely that a tidal barrage topped by a rail link would result in substantial economic growth at either end of the barrage, with industrial hubs supported by the transport link and the hydro-electrical power.

It would also reduce the contribution to reducing the overall contribution to achieving CO_2 emissions reductions targets, posing difficulties in assessing any positive environmental benefits yielded by a barrage scheme. In addition, the loss of the Severn Estuary would not be marginal loss of habitat, but rather the total loss of a unique ecosystem, raising challenges for the marginal valuation techniques of CBA (Section 4.7) and would represent a substantial loss to the natural capital assets of the UK.

The impact of the barrage schemes on ecosystem services in the Severn was estimated using a bundled approach to assessment, which assumes that one hectare of habitat provides all the services associated with that habitat, and that when one hectare of that habitat is lost, all of its services are lost. Such approaches can be used to inform about the amount of compensatory habitat that would be required to replace this loss. For scenario analysis, the high damage scenario assumes that all the services supported by the remaining habitat are lost, while the low damage scenario assumes the remaining habitat function as without loss.[425]

However, given the unique nature of the Severn Estuary, it is arguable that compensatory habitat suitable to fulfil the requirements of the Habitats Directive could not be created. Figure 33 shows the services considered in the assessment (dash: no service, filled circle: prominent service in this location, empty circle: provided but not primary). The impact of the Severn Barrage on the single ecosystem service of carbon sequestration was also explored, with calculations of the change in carbon emissions for each barrage option. The scheme is not being taken forward by the government, but if any individual scheme were adopted it would require an additional and more specific Environmental Impact Assessment to be undertaken.[425]

D5 Environmental Impact Assessment

EIA is undertaken 'down-stream' whereas SEA takes place 'up-stream' and evaluates the likely significant environmental impacts (including impacts on biodiversity) of a proposed project prior to a decision being made on development consent. SEA does not reduce the need for project-level EIA but it can help to streamline incorporation of environmental concerns (including biodiversity) into the decision-making process, often making project-level EIA a more effective process. EIA is intended to predict environmental impacts at an early stage in project planning and design and to find ways of reducing and possibly removing adverse impacts. For example, developers may be required to relocate flora and fauna to an alternative habitat.

Figure 33: Estimated Impacts of the Severn Barrage on Ecosystem Services[425]

Ecosystem services		FW Wet-lands	FW Rivers & Streams	Inter-tidal	Salt marsh	Sub-tidal	Terrestrial
PROVISIONING							
Food	Commercial fish catch (F)	-	●	○	-	●	-
	Shellfish catch (F)	-	-	○	-	○	-
	Grazing for cattle and sheep	-	-	-	●	-	○
	Subsistence level fishing & cropping (F)	○	○	○	○	○	○
	Wildfowling	-	-	●	●	-	●
Fibre/ materials	Fibre and construction prods (e.g. reeds, wood, leather, aggregates)	○	○	○	○	●	○
		-	-	-	-	-	●
		-	-	-	-	-	●
	Developed Land	-	-	-	-	-	●
Fuel	Renewable energy (F)	○	○	○	○	●	●
Water	Water for industrial usage	-	○	-	-	-	-
	Water for agricultural usage	○	●	-	-	-	-
	Regenerative services	○	○	○	●	●	●
	Maintenance of surface FW stores	○	○	-	-	-	-
	Groundwater replenishment	○	○	-	-	-	-
Natural medicines	Natural medicines	○	○	○	○	○	○
Biochemicals	Biochemicals and genetics	○	○	○	○	○	○
Ornamental resources	Ornamental resources	○	○	○	○	○	○
REGULATING							
Climate/climate change	C sequestration (F)	○	○	○	●	●	●
Air quality	Air quality	-	-	-	○	○	○
Water reg.	Flood protection	●	●	●	●	-	○
Water purification & waste management	Filtration of water	●	●	○	●	●	○
	Detoxification of water and sed.	●	●	●	●	●	-
Erosion regulation	Erosion regulation	-	○	●	●	●	●
Pollination	Habitat for bees	○	○	-	●	-	●
Bioremediation of waste	Beach cleaning? (F)	-	-	○	○	-	-

Ecosystem services		FW Wet-lands	FW Rivers & Streams	Inter-tidal	Salt marsh	Sub-tidal	Terrestrial
CULTURAL							
Spiritual, religious, cultural heritage	Archaeological ruins (F - historical not recreational value)	•	-	•	?	-	•
	Heritage fishing	-	•	-	-	•	-
Recreation and ecotourism	Freshwater angling (migratory)	-	•	-	-	-	-
	Freshwater angling (coarse)	-	•	-	-	-	-
	Estuarine & sea angling	-	-	•	-	•	-
	Bird watching (F)	•	•	•	•	-	•
	Hiking	•	•	•	•	-	•
	Diving	-	-	-	-	○	-
	Sailing, canoeing, surfing etc	-	•	-	-	•	-
	destination (incorporates views) (F)	•	•	•	•	•	•
	Archaeol. ruins (F - rec.value)	•	-	•	-	-	•
	Golf courses on dunes (F)	-	-	-	-	-	○
Landscape and amenity	Views (part of rec. above) AAF	•	•	•	•	•	•
	Visiting the Bore	-	-	-	-	•	-
SUPPORTING							
Soil form. & retention	Soil formation and retention	○	-	-	○	○	•
Cycling processes	Cycling processes	•	•	•	•	•	•
Primary production	Primary production	•	•	•	•	•	•
Habitat provision	Habitat provision	•	•	•	•	•	•

Projects require an EIA to meet the requirements of Council Directive 85/337/EEC on the assessment of the effects of certain public and private projects on the environment, as amended by Council Directive 97/11/EC. EIAs can stimulate public interest in planning decisions, as they are at the level at which individuals are directly affected, whereas SEAs are only of limited interest to the public and tend only to attract the attention of NGOs and similar stakeholders.

The statutory instrument applying the directive to the planning system in England (The Town and Country Planning (Environmental Impact Assessment) (England and Wales) Regulations 1999 (SI 293)) sets out the projects (major infrastructure projects) that require an EIA on a compulsory basis in Schedule 1 (Table 6). For certain other projects, which are listed in Schedule 2, planning authorities are required to consider the requirement for EIA.

A proposed development is a Schedule 2 development where it meets the criteria or exceeds the threshold set out in the schedule. The criteria and thresholds are not applied if a project is in, or partly within, a sensitive area (such as an SSSI), and all projects have to be screened to see if they are likely to have significant impacts on the environment by virtue of their size or location.

In general, agricultural projects fall outside the scope of the town and country planning system, except for those listed in schedule 2 (Table 7).Other land use projects are regulated under other consent procedures, including the Environmental Impact Assessment (Agriculture) (England) (No.2) Regulations 2006; the Environmental Impact Assessment (Land Drainage Improvement Works) (Amendment) Regulations 2005; and, the Environmental Impact Assessment (Forestry) (England and Wales) Regulations 1999 (SI 2228). There is a variety of other statutory instruments that relate to specific activities, such as marine fish farming and water abstraction, and sectors such as transport, energy and ports and harbours.

The EIA regulations are applied only to changes in the agricultural use of uncultivated and semi-natural land that affect more than 2ha, more than 4km of field boundaries or movements of 10,000m^3 or more of earth and rock. Semi-natural areas are defined by the plants and wildlife they support. The types of land considered semi-natural, such as species-rich hay meadow and unimproved grassland, are listed in the guidance notes of the EIA (agricultural) Regulations. The threshold is intended to reduce the administrative burden imposed on land managers, but has led to criticisms that there is continued loss of valuable habitat fragments in landscapes, which are often in patches of less than 2ha.[426] More widely, decisions on when an EIA is required under schedule 2 (Table 7), have led to a number of legal actions.[427]

Table 6: Activities Requiring an Environmental Impact Assessment under Schedule 1 of the Town & Country Planning (Environmental Impact Assessment) (England & Wales) Regulations 1999 That Always Require Environmental Impact Assessment (EIA) Irrespective of Their Location	
Number within Schedule 1	Description
Chemical & Petrochemical Industries	
Schedule 1 (1)	Crude oil refineries (excluding undertakings manufacturing only lubricants from crude oil) and installations for the gasification and liquefaction of 500 tonnes or more of coal or bituminous shale per day.
Schedule 1 (6)	Integrated chemical installations, for the manufacture on an industrial scale of substances using chemical conversion processes, where several units are juxtaposed and functionally linked to each other, and are for: The production of basic organic chemicals; The production of basic inorganic chemicals; The production of phosphorous, nitrogen or potassium based simple or compound fertilisers; The production of basic plant health products and biocides; The production of basic pharmaceutical products using chemical or biological processes; The production of explosives.
Schedule 1 (20)	Installations for the storage of petroleum, petrochemical or chemical products with a capacity of 200,000 tonnes or more.
Energy Industry	
Schedule 1 (2a)	Thermal power stations and other combustion installations with a heat output of 300 megawatts or more.
Schedule 1 (2b)	Nuclear power stations and other nuclear reactors (except research installations for the production and conversion of fissionable and fertile materials, whose maximum power does not exceed 1 kilowatt continuous thermal load).
Schedule 1 (3a)	Installations for the reprocessing of irradiated nuclear fuel.
Schedule 1 (3b)	Installations designed for: The production or enrichment of nuclear fuel; The processing of irradiated nuclear fuel or high level radioactive waste; The final disposal of irradiated nuclear fuel; The final disposal of radioactive waste; The storage (planned for 10 years or more) of irradiated nuclear fuels or radioactive waste in a different site from the production site
Extractive Industries	
Schedule 1 (14)	Extraction of petroleum and natural gas for commercial purposes where the amount extracted exceeds 500 tonnes per day for petroleum and 500,000 cubic metres for natural gas.
Schedule 1 (19)	Quarries and open-cast mining where the surface of the site exceeds 25 hectares, or peat extraction where the surface of the site exceeds 150 hectares.
Processing Industries	
Schedule 1 (4a)	Integrated works for the initial smelting of cast-iron and steel.
Schedule 1 (4b)	Installations for the production of non-ferrous crude metals from ore, concentrates or secondary raw materials by chemical, metallurgical or electrolytic processes.
Schedule 1 (5)	Installations for the extraction of asbestos and for the processing and transformation of asbestos and products containing asbestos: Asbestos cement products, with annual production of more than 20,000 tonnes of finished products; Friction material, with annual production of more than 50 tonnes of finished products; Other uses of asbestos where utilisation exceeds 200 tonnes per year.
Schedule 1 (18a)	Industrial plants for the production of pulp from timber or similar fibrous material
Schedule 1 (18b)	Industrial plants for the production of paper and board with a production capacity exceeding 200 tonnes per day.
Transport Infrastructure	
Schedule 1 (7a)	Construction of lines for long-distance railway traffic and of airports with a basic runway length of 2,100 metres or more.
Schedule 1 (7b)	Construction of motorways and express roads.
Schedule 1 (7c)	Construction of a new road of four or more lanes, or realignment and/or widening of an existing road of two lanes or less so as to provide four or more lanes, where such new road, or realigned and/or widened section of road would be 10 kilometres or more in a continuous length.

Schedule 1 (8a)	Inland waterways and ports for inland waterway traffic which permit the passage of vessels of over 1,350 tonnes.
Schedule 1 (8b)	Trading ports, piers for loading and unloading connected to land and outside ports (excluding ferry piers) which can take vessels of over 1,350 tonnes.
Schedule 1 (16)	Pipelines for the transport of gas, oil or chemicals with a diameter of more than 800 millimetres and a length of more than 40 kilometres.

Table 7: Agricultural Developments Requiring Consideration of the Need for an Environmental Impact Assessment Under Schedule 2 the Town & Country Planning (Environmental Impact Assessment) (England & Wales) Regulations 1999

Use of uncultivated or semi-natural land for intensive agricultural purposes	Development (such as greenhouses, farm buildings etc.) on previously uncultivated land is unlikely to require EIA unless it covers more than five hectares. In considering whether particular development is likely to have significant effects, consideration should be given to impacts on the surrounding ecology, hydrology and landscape.
Water management for agriculture, including irrigation and land drainage works	EIA is more likely to be required if the development would result in permanent changes to the character of more than five hectares of land. In assessing the significance of any likely effects, particular regard should be had to whether the development would have damaging wider impacts on hydrology and surrounding ecosystems. It follows that EIA will not normally be required for routine water management projects undertaken by farmers.
Intensive livestock installations	The significance or otherwise of the impacts of intensive livestock installations will often depend upon the level of odours, increased traffic and the arrangements for waste handling. EIA is more likely to be required for intensive livestock installations if they are designed to house more than 750 sows, 2,000 fattening pigs, 60,000 broilers or 50,000 layers, turkeys or other poultry.
Intensive fish farming	Apart from the physical scale of any development, the likelihood of significant effects will generally depend on the extent of any likely wider impacts on the hydrology and ecology of the surrounding area. Developments designed to produce more than 100 tonnes (dead weight) of fish per year will be more likely to require EIA.
Reclamation of land from the sea	In assessing the significance of any development, regard should be had to the likely wider impacts on natural coastal processes beyond the site itself, as well as to the scale of reclamation works themselves. EIA is more likely to be required where work is proposed on a site which exceeds one hectare

Several reviews of EIA have suggested that ecological resources outside protected sites tend to be inadequately assessed and evaluated.[428] The recent "Lawton Review" stated that environmental impact regulations do not provide sufficient protection for wildlife habitats, in particular small fragments. It also noted that the EU is currently reviewing these guidelines, and suggested the government should consider whether this gives an opportunity to provide better protection for remnant habitats.[57]

A case study evaluating EIA processes against the requirements of the ecosystems approach carried out for Defra identified a number of weaknesses, including that:[429]

- There is little consideration of ecosystem goods and services and their importance for human well-being.
- Future management of environmental assets are barely considered.
- The concepts of ecosystem functioning, thresholds and environmental limits do not form part of the impact assessment, which can narrowly focus on species and 'quality' of habitat.
- There is only limited valuation of environmental assets and at a late stage in the impact assessment.
- The public and stakeholder consultation processes within EIA are not as inclusive or as wide in scope as is required under the Ecosystems Approach.
- Cumulative impacts are generally poorly addressed and do not take ecosystem functions and processes into account.

Overall, the case study concluded that there were a number of options for incorporating the ecosystems approach into the EIA process if made explicit, but further regulation may be required to achieve this. The Royal Institute of Chartered Surveyors has also cited a number of relevant weaknesses in regard to EIA processes, including:[430]

- A tendency for considering the separate aspects of the environment (water quality, ecology, landscape etc.) in isolation without considering how they interact;
- inadequate assessment of the impacts of environmental changes on human well-being;
- difficulty in providing co-ordinated and enforceable mitigation beyond the project scheme completion date; and,
- inadequate consideration of cumulative impacts.

The EIA Directive requires development of mitigation measures to reduce the degree, extent, magnitude or duration of adverse environmental effects, but there is no absolute requirement to implement them. EU guidelines for the inclusion of cumulative impacts assessment for EIA of individual projects also exist, and consideration of cumulative environmental impacts is required under the EIA Directive. The Public Participation Directive 2003/35/EC also made amendments to the EIA Directive aimed at developing public participation and access to justice within EIA This has brought about some changes in how planning, land drainage and forestry projects are assessed. However, these fall short of the requirements of the ecosystem approach.

The current EIA approach to assessing impacts and identifying mitigation strategies is too reductionist, through narrowly focussing on individual species and environmental features.[431] The ecosystem approach requires consideration of processes, functions and the interaction between organisms (including humans) at multiple spatial and temporal scales, to ensure they are managed for the long term provision of ecosystem services. For an EIA to be used as part of an ecosystems approach, it would need to include greater consideration of likely long term and cumulative impacts on benefits arising from ecosystems in a given location, including:

- Systematic consideration and understanding by stakeholders and authorities of the likely impacts of a project on provision of ecosystem service benefits in the locality and how these will effect human well-being.
- Mitigation measures recommended for non-designated sites or species to conserve functioning ecosystems in the broader landscape, rather than just for the significant adverse impacts on species or habitats officially designated vulnerable or endangered, as at present.[432]
- Provision of evidence, or demonstrations, that mitigation measures can be implemented that will maintain levels of benefits from ecosystems in the locality.
- Guiding principles on what constitutes acceptable mitigation for loss of ecosystem services.

Environmental Statements

An Environmental Statement (ES) describes the relevant environmental information that has been obtained from the impact assessment and is required under Schedule 4 to the EIA regulations. The ES is required to describe the likely significant effects of the development on the environment and to specify environmental mitigation measures to avoid or reduce these. Legally binding commitments have to be secured under planning conditions.[433] The ES must be circulated to statutory consultation bodies and made available to the public for comment. Its contents, together with any comments, must be taken into account by the competent authority.

A recent report to Defra, [434] highlighted a number of issues with the assessment and implementation of environmental statements including frequent:

- failure to characterise ecological impacts properly;
- failure to mitigate for important ecological impacts properly; and,
- a lack of monitoring or follow up to ensure mitigation measures are appropriate and achieve required outcomes. [435]

Environmental Statements could be enhanced through the use of monetary values for ecosystem service benefits, Ecosystem Service Valuation (ESV), which could be used in CBA, and the development of alternative future scenarios resulting from a change in land or ecosystem use. Alternatively, decisions can be taken on the quantitative changes in ecosystem service benefits alone (the ESVA). This would be particularly applicable where changes could result in the breaching of Safe Minimum Standards for levels of ecosystem service provision within a given geographic area.

Ecological Impact Assessment

Ecological Impact Assessment (EcIA) is a tool that can be used in EIAs to identify, quantify and evaluate the potential impacts of defined actions on ecosystems or their components. If properly implemented, it could provide a scientifically defensible approach to ecosystem management,[435] but would need to be significantly extended in scope to incorporate consideration of ecosystem services.

The purpose of EcIA is to provide decision-makers with clear and concise information about the likely significant ecological effects associated with a project and to obtain the best possible biodiversity outcomes from land use changes. The Institute of Ecology and Environmental Management (IEEM) has set out guidelines on how EcIAs are conducted. An initial phase of the EcIA requires the identification of areas and resources that may be affected by the biophysical changes caused by the identified activities, however remote from the site, to identify the zone of influence. Once this is established the ecological resources affected are investigated and an ecological baseline determined for them.

The likely spatial and temporal limits of ecological impacts from this baseline for these resources are then surmised. There are various characteristics that can be used to identify ecological resources or features likely to be important and that require further investigation (if not already subject to statutory designation). These include:

- Animal or plant species, subspecies or varieties that are rare or uncommon, either internationally, nationally or more locally.

- Endemic species or locally distinct sub-populations of a species.
- Ecosystems and their component parts, which provide the habitats required by the above species, populations and/or assemblages.
- Habitat diversity, connectivity and/or synergistic associations (e.g. networks of hedges and areas of species-poor pasture that might provide important feeding habitat for rare species such as the greater horseshoe bat).
- Notably large populations of animals or concentrations of animals considered uncommon or threatened in a wider context.
- Plant communities (and their associated animals) that are considered to be typical of valued natural/semi-natural vegetation types. These will include examples of naturally species-poor communities.
- Species on the edge of their range, particularly where their distribution is changing as a result of global trends and climate change.
- Species-rich assemblages of plants or animals.
- Typical faunal assemblages that are characteristic of homogenous habitats.

Ecosystem Service Appraisal Mechanism

The EcIA would appear to be the most appropriate basis for incorporating further consideration of ecosystem services within EIAs, through inclusion of an assessment of ecosystem service benefit provision, commonly referred to as an Ecosystem Service Valuation Assessment (ESVA), as described in Chapter 4. This would involve the biophysical assessment of ecosystem services provided by a site, taking into account spatial considerations, such as how the site underpins key ecological processes within a wider geographical unit, such as at the river catchment or landscape level. For example, an area of semi-natural vegetation may support pollination services in a particular location.

The emphasis in the SEA and EIA Directives is on the prevention of damage to the environment through the development of alternative proposals in the pre-development phase and then the application of mitigation measures to limit or reduce the degree, extent, magnitude or duration of adverse effects. Appraisal mechanisms need to incorporate consideration of impacts on levels of ecosystem service benefit if they are to incorporate the ecosystem approach. This is both in terms of the likely impact of these changes on local communities and through incorporation of assessment of changes in benefits from ecosystems (Chapter 4).

However, in terms of implementation there is a lack of capacity in the planning system at the local authority level. Only about 35% of local planning authorities in England employ an ecologist or biodiversity officer,[434] and advice for planners regarding the interpretation and implementation of any system of ecosystem service assessments and trade-offs will be limited at relevant local scales. The capacity of Natural England to enforce and advise on planning applications and land use changes at the local scale is limited and is likely to be further reduced by public expenditure constraints.

D6 Marine Planning System

A separate marine planning system has been brought in under the Marine Act 2009, to implement "…a more strategic approach to managing marine activities and protecting marine resources in the future". This includes the production of Marine Policy Statements which will inform the production of ten marine spatial plans in UK waters. These will regulate the use of marine natural resources (excluding oil and gas) with the UK Economic Exclusion Zone (out to 200 nautical miles).

The plans for English territorial waters and Great Britain's offshore waters will be drafted by the Marine Management Organisation, created under the act. Plans for other UK territorial waters will be drawn up by the relevant devolved administrations. Under the act, powers to designate Marine Conservation Zones within territorial waters (out to 6 nautical miles) have also been introduced (POSTnote 310). These can exclude certain activities from areas of high biodiversity value, such as certain types of fishing, depending on the level of protection conferred.

It is not yet clear how this system will function in detail, although it is intended that the measures introduced under the Marine Act will fulfil the requirements of the EC Marine Strategy Framework Directive 2008/56/EC, including attaining "good environmental status" for marine areas (similar to good "ecological status" under the Water Framework Directive). Outside territorial waters (beyond 6 nautical miles), regulation of fishing activities falls under the Common Fisheries Policy (CFP). The CFP is currently under revision, and the revision is intended to include greater consideration of impacts on marine ecosystems in management of marine fisheries (POSTnote 357).

References

1 Haines-Young, R, Potschin, M, Cheshire, D, 2006, *Defining and Identifying Environmental Limits for Sustainable Development*. Final Report. Defra Project Code NR0102.
2 Tansley, A, 1935, *Ecology* 16: 284–307
3 Graves, A *et al*, 2009, *Valuation of Natural Resources: A NERC Scoping Study*.
4 Foley, J *et al*, 2005*, Science* 309: 570-574.
5 International Geosphere-Programme Biosphere http://www.igbp.net/page.php?pid=10
6 Rockstrom, J, *et al*, 2009, *Nature* 461: 472-475.
7 Crutzen, P, 2002 *Nature* 415: 23.
8 Milleninium Ecosystem Assessment (2005). *Ecosystems and Human Well Being: Synthesis* (Island Press, Washington, DC).
9 Carpenter S*, et al*, 2009, *PNAS* 106:1305–1312.
10 United Nations Environment Programme (UNEP), 2007, *Global environment outlook: environment for development, GEO 4*, UNEP/Earthprint: http://www.unep.org/geo/geo4/report/GE0-4_Report_Full_en.pdf.
United Nations World Water Assessment Program (UN WWAP), 2009, *UN World Water Assessment Programme 3rd UN World Water Development Report – Water in a Changing World (WWDR-3)*: http://unesco.org/water/wwap/wwdr/wwdr/wwdr3/pdf/WWDR3_Water_in_a_changing_World.pdf.
Organisation for Economic Co-operation and Development (OECD), 2008, *OECD Environmental Outlook to 2030*, OECD Publishing: http://www.oecd.org/dataoecd/29/33/40200582.pdf.
Intergovernmental Panel on Climate Change (IPCC), 2007, *The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, Cambridge University Press: http://www.ipcc.ch/ipccreports/ar4-wg1.htm.
International Assessment of Agricultural Knowledge, Science, and Technology for Development (IAASTD), 2009, *Agriculture at the Crossroads. The Global Report.* Island Press. Washington D.C: http://www.agassessment.org/reports/IAASTD/EN/Agriculture%20at%20a%20Crossroads_Global%20Report%20(English).pdf.
World Bank and Food and Agriculture Organisation, 2008, *The sunken billions: The economic justification for fisheries reform*, Agriculture and Rural Development Department, The World bank, Washington D.C: http://siteresources.worldbank.org/EXTARD/Resources/336681-1224775570533/SunkenBillionsFinal.pdf
11 Odum, E, 1971, *Fundamentals of ecology*, third edition, Saunders New York
12 Personal communication, Philip Bubb, WCMC-UNEP
13 Díaz S, 2006, *PLoS Biol* 4(8), 1300-1305
14 Speech by Professor John Beddington, UK Government Chief Scientific Adviser, http://www.govnet.co.uk/news/govnet/professor-sir-john-beddingtons-speech-at-sduk-09.
15 http://www.ost.gov.uk
16 Jackson, T, 2009, *Prosperity without Growth*, Sustainable Development Commission
Simms A, Johnson, V, Chowla, P, 2010. *Growth isn't Possible*, New Economics Foundation
17 Price, R, Durham, C, Chan, J, 2010, *Government Economic Service Review of the Economics of Sustainable Development*, Government Economic Service and Department for Environment, Food and Rural Affairs.
18 Linstead, C, 2008, *Reviewing Targets and Indicators for the Ecosystem Approach*, Final report Defra Project Code NR0119.
19 Royal Commission on Environmental Pollution, 1998, *Setting Environmental Standards*, Twenty-first Report.
20 http://www.cbd.int/ecosystem/
21 Sanderson, E, *et al*, 2002. BioScience, 52(10), 891-904
22 Millennium Ecosystem Assessment, 2005, *Ecosystems and Human Well Being:* Synthesis (Island Press, Washington, DC).
23 TEEB, 2010, *The Economics of Ecosystems and Biodiversity for National and International Policy Makers*, European Commission, Brussels. http://www.teebweb.org
24 Secretariat of the Convention on Biological Diversity, 2010, *Global Biodiversity Outlook 3*, Montreal, http://www.cbd.int/GBO3.
25 http://www.cbd.int/nagoya/outcomes/
26 http://ipbes.net/
27 Bartlemus, P, 2009, Ecological Economics, 68 (6)1850 – 1857.
28 Carpenter, S *et al*, 2009, PNAS 106, 1305-1312
Norgaard, R, 2009, Ecological Economics 69, 1219-1227
29 http://www.defra.gov.uk/evidence/science/how/documents/eis-100126.pdf
30 Anton, C, *et al*, 2010, Biodiversity and Conservation 19(10), 2979-2994
31 Adapted from Ecosystem Services for Poverty Alleviation, *Conceptual Framework for Analysis of Sustainable Management of Ecosystems to Maximise Poverty Alleviation*, http://www.nerc.ac.uk/research/programmes/espa/documents/Final%20Report%20China%20-%20annex.pdf
32 *Pers. comm.*, Ollerton J, University of Northampton, 2009.
33 *Pers. comm*., Potts S, Reading University, 2009.
34 *Pers. comm*., Potts S, Reading University, 2009.
35 Ricketts T, *et al,* 2008, *Ecology Letters* 11, 499-515.
36 Memmott J, *et al.,* 2004, *Proc Roy Soc B* 271, 2605-2611.
37 Carvell, C, *et al*, 2007, *Journal of Applied Ecology*, 44 (1), 29-40
38 Petchey, O, Gaston, K, 2006, *Ecology Letters*, 9, 741–758
39 Luck, G, *et al*, 2009, *Bioscience*, 59(3), 223-235
40 Haslett, J, *et al*, 2010, *Biodiversity and Conservation*, 19 (10), 2963-2977
41 Kremen, C, 2007, *Ecology Letters*, 10, 299-314.
42 Hougner, C, Colding, J, Soderquist, T, 2006, *Ecological Economic* 59, 364-374
43 Mols, C, Visser, M, 2007, *PLoS One* 2,e202.
44 Slingenberg, A, *et al*, 2009, *Study on understanding the causes of biodiversity loss and the policy assessment framework*, Report to the European Commission Directorate-General for Environment by ECORYS Research and Consulting.
45 Secretariat of the Convention on Biological Diversity, 2010, Global Biodiversity Outlook 3, Montréal, 94 pages. http://www.cbd.int/GBO3.
46 Imhoff, M, *et al*, 2004, *Nature*, 429, 870-873
47 Bai, Z, Dent, L, Olsson L, Schaepman, M, 2008, *Global assessment of land degradation and improvement. 1. Identification by remote sensing*. Report 2008/01, ISRIC – World Soil Information, Wageningen
48 UN *Fourth Global Environment Outlook* (GEO-4).
49 Braat & ten Brink 2008, *The cost of policy inaction*, EU study

50 Balmford, A, *et al*, 2003, *PNAS* 100, 1046-1050
51 IUCN. 2009. *Red List of Threatened Species.*
52 Imhoff, M, *et al*, 2004, *Nature* 429, 870-873
Haberl, H, *et al,* 2007, *PNAS* 104, 12942-12947
53 http://www.diversitas-international.org/docs/DIVERSITAS%20OSC2%20First%20press%20release.pdf
54 European Academies Science Advisory Council, 2009, *Ecosystem Services and Biodiversity in Europe*, EASAC Policy Report 09
55 De Heer, M, Kapos, V, ten Brink, B, 2005, *Philosophical Transaction of the Royal Society B* 360, 297-308
56 Maclean, N, 2010, *Silent Summer: the state of wildlife in Britain and Ireland*, Cambridge University Press: Cambridge.
57 Lawton, J, *et al*, 2010, *Making Space for Nature: a review of England's wildlife sites and ecological network*, Report to Defra.
58 Defra 2002. *England Biodiversity Strategy*.
59 Biodiversity Action Reporting System (BARS), http://ukbap-reporting.org.uk
60 JNCC, 2010, *UK Biodiversity Action Plan: Highlights from the 2008 reporting round*, http://www.jncc.gov.uk/pdf/pub2010_UKBAPHighlightsReport2008.pdf
61 Preston, C *et al*, 2002, *The Changing Flora of the UK*. Defra.
62 Countryside Survey, 2009, *Countryside Survey: England Results from 2007*, NERC/Centre for Ecology and Hydrology, Defra, Natural England, 119pp. (CEH Project Number:C03259).
63 Natural England, 2008, *State of the Natural Environment*, Sheffield, Natural England.
64 Morecroft, M, et al, 2009, *Biological Conservation*, 142 (12). 2814-2832.
65 Robinson, R, 2010, *State of bird populations in Britain and Ireland*, In: Maclean, N, (Ed.), Silent Summer: the state of wildlife in Britain and Ireland. (pp. 218-318), Cambridge University Press: Cambridge.
66 Halliday, T, 2010, *Amphibians*, In: Maclean, N, (Ed.). Silent Summer: the state of wildlife in Britain and Ireland, (pp. 363-382) Cambridge University Press: Cambridge
67 Gordon, L, Petersen, G, Bennett, E, 2008, Trends Ecol Evol, 23 (4), 211-9
68 Lavorel S, *et al*, 2010, *the RUBICODE Project. Rationalising Biodiversity Conservation in Dynamic Ecosystems*, Funded under the European Commission Sixth Framework Programme.
Tyliankis, J, et al. 2008. *Ecology Letters*, 11, 1351-1363.
69 Nystrom, M, *et al*, 2008, *Coral Reefs* 27(4): 795-809.
70 Carpenter, S, 2005. *PNAS* 102, 10002-10005.
71 Haines-Young, R, Potschin, M. 2009, *Environmental Limits, Ecosystem Resilience and Supporting Services*, Pilot Review, Report for Defra
72 Biggs, R, Carpenter, S, Brock, W, 2009, *PNAS* 106 (3) 826-831.
73 Scheffer, M, *et al*, 2009, Nature 461, 53-59
74 Ludwig, D, *et al*, 1997, *Landscape Ecology Function and Management Principles from Australia's Rangelands*, Collingwood, Australia: CSIRO.
Walker, B, *et al*, 1981, *J. Ecol.* 69, 473-98.
75 Savory, A, 1998, *Holistic Management: A New Decision Making Framework*, Second edition, Island Press
Sankraran, M, *et al*, 2005. *Nature* 438, 846-849.
76 Carpenter, S, Westley, F, Turner, M, 2005, *Ecosystems* 8 (8): 941-944.
77 Biggs, R, Carpenter, S, Brock, W, 2009, *PNAS*, 106 (3) 826-831.
78 TEEB, 2010, *The Economics of Ecosystems and Biodiversity for National and International Policy Makers*, European Commission, Brussels. http://www.teebweb.org
79 Rockström, J, 2009, *Nature*, 461, 472-475
80 Ekins, P, *et al*, 2003, Ecological Economics, 44, 165–185
81 TEEB, 2010, *The Economics of Ecosystems and Biodiversity for National and International Policy Makers*, European Commission, Brussels. http://www.teebweb.org
82 http://www.globeinternational.info/wp-content/uploads/2010/09/Nagoya-Declaration-on-Parliamentarians-and-Biodiversity-26-10-10-FINAL.pdf
83 Globe International Commission on Land Use Change and Ecosystems, 2010, *Natural Capital Action Plan*
84 Adapted from Daily, G, Turner, R, 2008, *Environ. Resource Econ* 39,25-3
85 Haines-Young, R, Potschin, M, 2009, *Methodologies for defining and assessing ecosystem services*, Final Report, JNCC
Boyd, J, Banzhaf, S, 2007, *Ecological Economics* 63, 616-626.
Constanza, R, 2008, *Biological Conservation* 141,350-352
Wallace, K, 2007, *Biological Conservation* 139, 235 -246
Fisher, B, Turner, K, 2008, *Biological Conservation* 141,1167-1169
Fisher, B, Turner, R, Morling, P, 2009, *Ecological Economics* 68(3), 643-653.
86 Wallace, K, 2007, *Biological Conservation* 139, 235 -246.
87 Fisher, B, Turner, R, Morling, P, 2009, *Ecological Economics*, 68 (3), 643-653
88 Fisher, B, Turner, R, 2008, *Biological Conservation*, 141, 1167-1169
89 http://www.valuingthearc.org/
90 Constanza, R, 2008, *Biological Conservation* 141:350-352
91 Haines-Young, R, Potschin, M, 2009, Methodologies for defining and assessing ecosystem services, Final Report, JNCC, Project Code C08-0170-0062, 69 pages
92 Balmford, A, *et al*, 2008, *The Economics of Biodiversity and Ecosystems: Scoping the Science*, Cambridge, UK, European Commission
93 Mäler, K-G, Aniyar, S, Jansson, A, 2009, *Environmental Resource Economics*, 42, 39-51
94 Boyd, J, Banzhaf, S, 2007, *Ecological Economics* 63, 616-626
95 TEEB, 2010, The Economics of Ecosystems and Biodiversity for National and International Policy makers, Chapter 3 – Strengthening Indicators and Accounting Systems for Natural Capital
96 Weber, J.-L, 2007. *Ecological Economics*, 61, No4, 708-715.
97 Lavelle, P, et al, 2006, *European Journal of Soil Biology*, 42 (Supplement 1) Pages S3-S15
98 Royal Commission on Environmental Pollution, 2007, *The Urban Environment*, Twenty-Sixth Report
99 Huguenin, M; Leggett, C; Paterson, R, 2006, *European Journal of Soil Biology*, 42 (Supplement 1) Pages S16-S22
100 ADAS, 2006, *Economic Valuation of Soil Functions Phase 1: Literature Review and Method Development*, Report to Defra.
101 Bradley, R, 2005, *Soil Use and Management*, 21, 363-369
102 Dawson, J; Smith, P, 2007, *Science of the Total Environment,* 382*, 165-190.*
103 Thompson, D, 2008, Carbon Management by Land and marine managers: review of the evidence, Natural England report
104 European Environment Agency, 2010, *Ecosystem accounting and the cost of biodiversity losses: The case of coastal Mediterranean wetlands*, ISSN Technical report series 1725-2237.

105 Haines-Young, R, Potschin, M, 2010, *Proposal for a Common International Classification of Ecosystem Goods and Services (CICES) for Integrated Environmental and Economic Accounting*, Report to the European Environment Agency
106 Bartelmus, P, 2009, *Ecological Economics* 68(6): 1850-7
Maler, K.G., Aniyar, S., Jansson, A. 2009, *Environmental and Resource Economics* 42(1): 39-51.
107 Haines-Young, R, Potschin, M, 2009, *Methodologies for defining and assessing ecosystem services*, Final Report, JNCC, Project Code C08-0170-0062, 69 pages
108 From Haines-Young, R, 2009, *Environmental Limits, Ecosystem Resilience and Supporting Services*, Pilot Review Report for Defra.
109 Adapted from http://www.defra.gov.uk/evidence/economics/foodfarm/reports/envacc/index.htm
110 TEEB, 2010, *The Economics of Ecosystems and Biodiversity for National and International Policy Makers, Chapter 1,*
111 Ekins, P, *et al*, 2003, Ecological Economics, 44, 165–185
Ekins, P, Simon, S, 1998, *Determining the sustainability gap: national accounting for environmental sustainability*, In: Vaze, P. (Ed.), UK Environmental Accounts: Theory, Data and Application, Office for National Statistics, London
Ekins, P, Simon, S, 1999, *International Journal of Environment and Pollution* 2 (1), 24-58
Ekins, P, Simon, S, 2001, *Ecological Economics* 37 (1), 5-22
112 Mäler, K-G, Aniyar, S., Jansson, A, 2009, *Environmental Resource Economics* 42: 39-51
Bartelmus, P, 2009, Ecological Economics 68(6): 1850-1857
113 Defra, 2007, *Securing a healthy natural environment: an action plan for embedding an ecosystems approach*, www.defra.gov.uk/wildlife-countryside/pdf/natural-environ/eco-actionplan.pdf
114 Maltby, E, 1997, *Ecosystem Management: The concept and the strategy,* IUCN World Conservation 3/97.
115 Anderson, B, 2009, *Journal of Applied Ecology* 46, 888-896.
116Smart, S, *et al*, 2010, *An Integrated Assessment of Countryside Survey data to investigate Ecosystem Services in Great Britain*, CS Technical Report 10/07, Centre for Ecology and Hydrology
117 Dixon, M *et a*l, 2008, *Maritime Engineering*, 161(2), 61-71.
118 TEEB, 2010, *The Economics of Ecosystems and Biodiversity for National and International Policy Makers*.
119 Maltby, E, *et al*, 1999, *Ecosystem Management: Questions for science and society*, Royal Holloway Institute for Environmental Research.
120 Sniffer, 2010, *The Use of Environmental Limits in Regulating Environmental Systems – How could the concept be applied to Environmental Agencies*? Project UKCC14.
121 Government Office for Science, 2010, *Land Use Futures: Making the most of Land in the 21st Century*, Final Project Report
122 The House of Commons Environmental Audit Committee, Thirteenth Report of 2007-08 Session, *Halting Biodiversity Loss*.
123 Pethick, J, 2001, *CATENA* 42, 307-322.
124 Dixon, M, Leggett, D, Weight, R, 1998, *Journal of the Chartered Institution of Water and Environmental Management* 12, 107-112.
125 Everard, M, Watts, B, 2009, *Ecosystem services case studies*, Environment Agency Better Regulation Science Programme Report.
126 Cave, R, *et al*, 2003, *The Science of The Total Environment* 314, 31-52.
127 Turner, R, *et al*, 2007, *Global Environmental Change* 17, 397-407.
Luisetti, T, *et al*, 2008, *An Ecosystem Services Approach for Assessing Managed Realignment Coastal Policy in England*, CSERGE Working Paper
128 http://www.nottingham.ac.uk/cem/pdf/NEA_scenario_stroylines_V1%20draft_13_9_2010.pdf
129 Sala, O, *et al*, 2000. *Science*, 287, 1770-1774.
130 http://uknea.unep-wcmc.org/
131 Watson, R, Albon, S, 2010, *UK National Ecosystem Assessment. Preliminary Synthesis and Progress Report on Status and Trends*, UK NEA
132 National Audit Office, 2009, *Delivering High Quality Impact Assessments*, Report by the Comptroller and Auditor General, HC 128 Session 2008-2009
133 Cowling, R, *et al*, 2008, *PNAS*, 105 (28): 9483–9488
134 Haines-Young, R, Potschin, M, 2008, England's *Terrestrial Ecosystem Services and the Rationale for an Ecosystem Approach*, Full Technical Report. Defra Project Code NR0107
Haines-Young, R, Potschin, M, 2009, *Methodologies for defining and assessing ecosystem services*, Final Report, JNCC, Project Code C08-0170-0062.
135 Smart, S, *et al*, 2010*, An Integrated Assessment of Countryside Survey data to investigate Ecosystem Services in Great Britain*, CS Technical Report 10/07
136 Paterson, J, et al, 2010, *Scenario Narratives for the UK NEA*, CEM working Paper No 5
137 Reid, W, *et al*, 2006, *Bridging scales and knowledge systems: concepts and applications in ecosystem assessment / Millennium Ecosystem Assessment*
138 http://www.finding-sanctuary.org/
139 Haines-Young, R, Potschin, M, 2009, *Methodologies for defining and assessing ecosystem services*, Final Report, JNCC
Chan, K, *et al*, 2006, *PLoS Biol* 4, 2183-2152
Naidoo, R, *et al*, 2008, *PNAS* 1059 (28) 9495-9500
Imhoff, M, *et al*, 2004, *Nature* 429 (6994), 870-873
Schröter, D, et al, 2005, *Science* 310 (5752), 1333-1337
140 Daily, G., Matson, P, 2008, *PNAS* 105(28): 9455-9456
141 Haines-Young, R.H. Potschin, M.B. 2009. *Methodologies for defining and assessing ecosystem services*. Final Report, JNCC
Daily, G, et al, 2009, *Frontiers in Ecology and Environment* 7(1), 21-28
142 Brown, T, Bergstrom, J, Loomis, J, 2007, *Natural Resources Journal* 47 (2), 329-376.
143 O'Donnell, G, 2008, *Multiscale experimentation, monitoring and analysis of long-term land use changes and flood risk*, Proceedings of the 10th National Hydrological Symposium, Exeter.
Stewart, A, Lance, A, 1983, *J. Environ. Manage*.17,18-99
Holden, J et al, 2006, *J. Environ. Manage*, 35,1764-1778
Heathwaite, A., Burt, T.Trudgill, S. 1990. *Land-use controls on sediment production in a lowland catchment, south-west England*. In: Soil Erosion on Agricultural Land, Boardman, J., Forster, I., Dearing, J. (eds). John Wiley and Sons Ltd. Chichester.
144 Clark, J, *et al*, 2010, *Climate Research*, 45, 131-150
145 McGrath, M, 2008, *United Utilities Sustainable Catchment Programme Update November 2008*, Presentation to CAR conference.
146 Watson, R, Albon, S, 2010, *UK National Ecosystem Assessment, Preliminary Synthesis and Progress Report on Status and Trends,* UK NEA
147 Rural Economy and Land Use Programme, 2010, *Integrated Management of Floodplains*, Policy and Practice Note 15.
148 Posthumus, H, *et al*, 2009, *Journal of Flood Risk Management*, 2,1-8
149 Graves, A, *et al*, 2009, *Valuation of Natural Resources: A NERC Scoping Study*
150 Graves, A, *et al*, 2009, *Valuation of Natural Resources: A NERC Scoping Study*
Pagiola, S, Von Ritter, K, Bishop, J, 2004, *Assessing the Economic Value of Ecosystem Conservation*, TNC-IUCN-WB, Washington DC
151 Haines-Young, R, Potschin, M, 2009, Methodologies for defining and assessing ecosystem services, Final Report, JNCC

Brown, T, Bergstrom, J, Loomis, J, 2007, *Natural Resources Journal* 47(2),329-376
Chee, Y, 2004, *Biological Conservation* 120(4), 549-565.
Farber, S et al, 2006, *Bioscience* 56(2):121-133.
Graves, A *et al*, 2009, *Valuation of Natural Resources: A NERC Scoping Study*.
Pagiola, S, Von Ritter, K, Bishop, J, 2004, *Assessing the Economic Value of Ecosystem Conservation*, TNC-IUCN-WB, Washington DC.
152 Clark, J, Burgess, J, Harrison, C, 2000, *Ecological Economics*, 33, 45-62.
153 Adapted from Haines-Young, R, Potschin, M, 2009, *The links between biodiversity, ecosystem services and human well-being*, Ch 7 in Raffaelli, D, and Frid, C, Ecosystems Ecology, a new synthesis, BES ecological reviews series, Cambridge University Press.
154 Ozdemiroglu E, *et al*, 2009, *Land Use Futures Environmental Valuation Paper For Foresight*, Eftec
155 Government Office for Science, 2010, *Land Use Futures: Making the most of Land in the 21st Century*, Final Project
156 European Academies Science Advisory Council, 2009, Ecosystem services and biodiversity in Europe, EASAC policy report 09.
157 Luck, G.W. et al. 2003. Population diversity and ecosystem services. Trends in Ecology and Evolution 18: 331-336.
158 Kremen, C. 2005. Managing ecosystem services what do we need to know about their ecology? Ecology Letters 8, 468-479.
159 Price, R., Durham, C. and Chan, JY.2010. Government Economic Service Review of the Economics of Sustainable Development. Government Economic Service and Department for Environment, Food and Rural Affairs.
160 Globe International Commission on Land Use Change and Ecosystems, 2010, *Natural Capital Action Plan*

162 Ozdemiroglu E, et al, 2009, Land Use Futures Environmental Valuation Paper For Foresight, Eftec.
162 Potschin, M, Haines-Young, R, 2006, *LUP* 75 (3-4), 621-174
163 Fish, R, 2010, *Participation and an ecosystems approach to decision making*, Draft Guidelines NR0124
164 Reed, M, 2008, *Biological conservation*, 141 (10), 2417-2431
165 Defra, 2007, *Public understanding of the concepts and language around ecosystem services and the natural environment* Http://www.defra.gov.uk/environment/policy/natural-environ/research/programme.htm
166 Glaves, P, Egan, D, 2010, *Bulletin of the Institute of Ecology and Environmental Management*, 68, 12 -15
167 Reed, M, et al, 2009, *Journal of Environmental Management*, 90, 1933-1949
168 Everard, M, et al, 2009, *Environmental Law and Management* 21, 76-82.
169 OECD, 2004, *Recommendation of the Council on the use of economic instruments in promoting the conservation and sustainable use of biodiversity*
170 Dasgupta, P, 2008, *Environmental and Resource Economics*, 39, 1-7.
Weitzman. M, 2009, *The Review of Economics and Statistics* XCI, 1-19.
171 RELU, 2009, *Catchment management for the protection of water resources: The Ecosystem Health Report Card*, Policy and Practice Notes 7.
172 RELU. 2009, *Catchment management for the protection of water resources: The Ecosystem Health Report Card,* Policy and Practice Notes 7.
173 Formally the social discount rate (r) is defined as: $r = p+\mu.g$ where (p) is the rate at which individuals discount future consumption over present consumption, (μ) elasticity of the marginal utility of consumption and (g) annual growth in per capita consumption. The 2003 HM Treasury Green Book details the basis of the discount rate recommended for use in the appraisal of projects and policies as: r = 3.5%, based on p=1.5%; μ =1.0 and g=2%.
174 Defra, 2009, *Marine and Coastal Access Bill Impact Assessment: Introduction to the House of Commons*, London, Defra
175 Fisher, B, *et al*, 2008, *Ecological Applications*, 18(8), 2050-2067
176 Turner, R, 2007, *Environ. Resource Econ*, 37,253-269.
177 Diagram adapted from Potschin, M, Haines-Young, R, 2008, *Limits, thresholds and the sustainability choice space*, In: Sustainability Impact Assessment of Land Use Changes. Edited by Helming, K., Perez-Soba,M., Tabbush, P.
178 White, P, Pickett, S, 1985, *Natural disturbance and patch dynamics: An introduction*, pp.3-13, In: S, Pickett and P, White editors, The ecology of natural disturbance and patch dynamics. Academic Press, Orlando.
179 Scott, C, *et al*, 2009, *Assessing the resilience of ecosystem delivery: Can the Earth bounce back, Review for Defra,* NERC and LWEC
180 Petchey, O, Gaston, K, 2006, *Ecology Letters*, 9, 741–758
181 Hughes, A, Stachowicz, J, 2004, *Proceedings of the National Academy of Sciences of the United States of America*, 101 (24): 8998-9002.
182 Haines-Young, R., Potschin, M, 2010, *The Resilience of Ecosystems to Environmental Change (RECCE)*, Full Technical Report, 129 pp. Defra Project Code: NR0134
183 Van Ruijven, J, Berendse, F, 2010, *Journal of Ecology*, 98, 81-86.
184 Isbell, F, Polley, F, Wilsey, B, 2009, *Ecology Letters* 12: 443-451
Isbell, F, Polley, F, Wilsey, B, 2009, *Ecology Letters* 12: 443-451.
185 Bennett, E, Petersen, G, Gordon, L, 2009, *Ecology Letters*, 12, 1-11.
186 Scheffer, M, Carpenter, S, 2003, *Trends in Ecology and Evolution*, 18, 648-656.
187 Hughes, et al, 2007, *Curr. Biol*., 17: 360-365.
188 Enfors, E, Gordon, L, 2007, *Land Degradation and Development* ,18, 680-696
189 F.S.Chapin *et al* (eds), 2009, *The principles of Ecosystem Stewardship*, Springer Science
190 Adger, W, *et al*, 2005, Science, 309, 1036-1039
191 International Council for Science. 2002. ICSU Series on Science for Sustainable Development: Resilence and Sustainable Development. No 3.`
192 Dawson, T, et al, 2010, *Biodiversity and Conservation*, 19, 2843-2853
193 Gomez-Baggethun, E, Olsson, P, Montes, C, 2009, *Learning with Crises: Building Resilience to Secure Ecosystem Services in Mediterranean Resource Management Systems*, Proceedings of the 8th International Conference of the European Society for Ecological Economics
194 Dawson, T, *et al*, 2010, *Biodiversity and Conservation*, 19, 2843-2853
195 Bennett, E, Petersen, G, Gordon, L, 2009, *Ecology Letters*, 12, 1-11
Del-Val, E, *et al*, 2006, *Ecosystems*, 9, 598-608
196 Ricketts, T, *et al*, 2008, *Ecology Letters*, 11, 499-515.
197 Adapted from Dawson, T, *et al*, 2010, *Biodiversity and Conservation*, 19, 2843-2853
198 Bennett, E, Petersen, G, Gordon, L, 2009, *Ecology Letters*, 12, 1-11
199 Agrawala, S, Fankhauser, S, (eds), 2008, *Economic aspects of adaptation to climate change: costs, benefits and policy instruments*, OECD, Paris.
200 Adger, W, *et al*, 2009, Climatic Change, 93, 335-354
201 Baker, A, 1992, *Introduction: on ideology and landscape*, In: Baker, A., Biger, G. (eds) Ideology and landscape in historical perspective, Cambridge University Press, Cambridge.
202 Folke, C, *et al*, 2002, *Resilience and sustainable development: building adaptive capacity in a world of transformations*, The Environmental Advisory Council to the Swedish Government Scientific Background Paper
203 Ostrom, E, 2009, *Science*, 325, 419-422
204 Dawson, T, *et al*, 2010, *Biodiversity and Conservation*, 19, 2843-2853
Waldrop, M, 1994, *Complexity: the emerging science at the edge at the of order and chaos*, Penguin, Harmondsworth, UK

205 Dawson, T, *et al*, 2010, *Biodiversity and Conservation*, 19, 2843-2853
Corning, P, 2002, *Complexity*, 7(6),18-30
Kauffman, S, 1993, *The origins of order: self-organisation and selection in evolution*, Oxford University Press, New York.
206 Young, O, *et al*, 2006, *Global Environmental Change*, 16, 304-316
207 Adger, W, *et al*, 2009, *Climatic Change*, 93, 335-354
McIntosh, R, Tainter, J, McIntosh, S, (eds), 2000, *The way the wind blows: climate, history and human action*, Cambridge University Press, Cambridge.
208 Eakin, H., *et al*, 2009, *Hidden costs and disparate uncertainties: trade-offs involved in approaches to climate policy*, In: Adger, W, Lorenzoni, I, O'Brien, K, (eds), Adapting to climate change: thresholds, values, governance, Cambridge University Press, Cambridge.
209 Mäler K, Li, C-Z, Destouni, G, 2007, *Pricing resilience in a dynamic economy environment system: A capital theoretical approach*, Beijer Discussion Paper Series no.208 (Beijer Institute, Stockholm).
210 Mäler K, Anlyar, S, Jansson, Ä, 2008, *PNAS* 105 (28), 9501-9506
211 Haines-Young, R, Potschin, M, (2007), *The Ecosystem Concept and the Identification of Ecosystem Goods and Services in the English Policy Context*, Review Paper to Defra, Project Code NR0107
212 Philip Bubb, IUCN, personal communication
213 Lock, K, 2010, IEEM Conference Proceedings
214 Wheeler, B, *et al*, 2004, *Eco-hydrological Guidelines for Lowland Wetland Plant Communities*, Eds Brooks, A, José, P, Whiteman, M, Peterborough, Environment Agency
215 NERC funded project NE/F009232/1, *Impact of summer flooding on floodplain biodiversity via nutrient deposition*, Prof David Gowing, Open University.
216 Pywell *et al*, 2002, *Journal of Applied Ecology*, 39, 294-309
217 Harris, J, 2009, *Science*, 325, 573-574
218 Kustler, J, Kentula, M, 1990, *Wetland Creation and Restoration: The Status of the Science,* Island Press, Washington D.C
219 Banayas, J, *et al*, 2009, *Science,* 325, 1121 – 1124.
220 Hobbs, R, Jackson, S, 2009, *Science*, 325, 567-568
221 Palmer, M, Filoso, S, 2009, *Science*, 325, 575 -576
222 Pywell, R, 2010, NERC Centre for Ecology and Hydrology - Pers Comm.
223 Kettunen, et al, 2007. *Guidance on the maintenance of landscape connectivity features of major importance for wild flora and fauna - the implementation of Articles 3 and 10 of the Birds and Habitats directives*, Institute for European Environmental Policy.
Donald, P, Evans, A, 2006, *Journal of Applied Ecology* 43, 209-218.
Bailey, S, 2007, *Forest Ecology and Management* 238, 7-23
224 Moilanen, A, *et al*, 2005, *Proceedings of the Royal Society B*, 272, 1885-1891
225 Gonzalez, A. *et al*, 1998, *Science*, 281, 2045-2047.
226 Catchpole, R, Buchanan, K, 2007, *Adapting to climate change – 10 steps to heaven, Practical steps in assisting the natural environment to cope with climate change*, Unpublished Repor, Natural England, Peterborough.
227 Original figure produced by Catchpole R. in Lawton, J et al, 2010, *Making Space for Nature: a review of England's wildlife sites and ecological network*, Report to Defra.
228 Kleijn, D, *et al*, 2009, *Proceedings of the Royal Society B-Biological Sciences*, 276 (1658), 903-909, 1
229 Woodcock, B, *et al*, 2008, *Oecologia*, 154, 773-783
Bradbury, R, *et a*l, 2008, *Bird Study*, 55:23-31
230 Critchley, C, Burke, M, Stevens, D, 2003, *Biological Conservation*, 115, 263-278
Hewins, *et al*, 2005, *The condition of lowland BAP priority grasslands: results from a sample of non-statutory stands in England*, English Nature Research Report No.636, English Nature: Peterborough.
231 De Bello, F, *et al*, 2009, *Biological Conservation*, 143, 9-17
232 Rollett, A., *et al*, 2008, *Delivering environmental services through agri-environment programmes: a scoping study*, Report to Land Use Policy Group
233 Lactacz-Lohmann, U, Schilizzi, S, 2005, *Auction for conservation contracts: A review of the theoretical and Empirical literature*, Report to Scottish Executive Environment and Rural Affairs Department
234 http://www.nceas.ucsb.edu/projects/9381
235 Rouquette, J, 2009, *Journal of Applied Ecology*, 46(2), 289-296.
236 Environmental Law Institute, 2007, *Wetland Mitigation Banking*, Environmental Law Institute, Washington D.C
237 Anderson, B, *et al*, 2009. *Journal of Applied Ecology*, 46, 888-896.
238 Treweek, J, 2009, *Scoping study for the design and use of biodiversity offsets in an English Context*, Final report to Defra.
239 Everard, M, Appleby, T, 2009, *Environmental Law and management*, 21, 14 -21
240 Defra, 2010, *Payments for Ecosystem services, A short introduction*
241 Wunder, S, 2005, *Payments for Environmental Services, Some nuts and bolts*, CIFOR Occasional Paper No 42
242 Perrot-Maitre, D, 2006, The *Vittel payments for ecosystem services: a perfect PES case?* International Institute for Environment and Development report for DFID, London, UK
243 Gunderson, L, Peterson, G, Holling, C, 2008, *Practicing Adaptive Management in Social-Ecological Systems. In Complexity Theory for a Sustainable Future*, Edited by Jon Norberg and Graeme Cumming Columbia University Press
244 International Council for Science, 2002, *ICSU Series on Science for Sustainable Development: Resilence and Sustainable Development*, No 3.
245 Peterson, G, 2009, *Ecological limits of adaptation to climate change. In: Adapting to Climate Change: Thresholds, Values, Governance*, eds. W. Neil Adger, Ireni Lorenzoni and Karen L. O'Brien, Published by Cambridge University Press 2009
Gordon, L, Peterson, G, Bennett, E, 2007, *Trends in Ecology and Evolution*, 23(4), 211-219
246 International Council for Science, 2002, *ICSU Series on Science for Sustainable Development, Resilence and Sustainable Development*, No 3.
247 Gunderson, L, Peterson, G, Holling, C, 2008, *Practicing Adaptive Management in Social-Ecological Systems. In Complexity Theory for a Sustainable Future*, Edited by Jon Norberg and Graeme Cumming Columbia University Press
248 Adger, W, *et al*, 2009, *Climatic Change*, 93, 335-354
249 Berkes, F, Colding, J, Folke, C, 2002, *Navigating Social Ecological Systems: Building Resilience for Complexity and Change*, Cambridge University Press, Cambridge.
250 Ayensu E, *et al*, 1999, *Science*, 286, 685–686
251 Resilience Alliance, 2007, *Assessing Resilience in Social-Ecological Systems,* A Workbook for Scientists, Version 1.1
Carpenter, S, *et al*, 2009, *PNAS*, 106 (5): 1305-1312.
252 Mitchell, R *et al*, 2000, *Evolution and Systematics*, 32, 142-160.
253 Pielke Jr, R, 2007,*The Honest Broker: Making Sense of Science in policy and Politics*, Cambridge University Press, Cambridge.
254 Pielke, Jr, R, 2007, *The Honest Broker: Making Sense of Science in Policy and Politics*, Cambridge University Press, Cambridge.
255 http://www.bis.gov.uk/assets/biscore/goscience/docs/g/10-669-gcsa-guidelines-scientific-engineering-advice-policy-making.pdf

256 Lempert, R, 2006, *Manag. Sci*. 52, 514-528
257 Eakin, H, *et al*, 2009, *Hidden costs and disparate uncertainties: trade-offs in approaches to climate policy*, In: Adapting to Climate Change: Thresholds, Values, Governance, eds, W, Neil Adger, Ireni Lorenzoni and Karen L. O'Brien. Published by Cambridge University Press.
258 Kahan, D, 2010, *Nature*, 463, 296-297.
Kahan, D, *et al*, 2007, *Journal of Empirical Legal Studies,* 4, 465-505
259 Ostrom, E, 2005, *Understanding institutional diversity*, Princeton University Press, Princeton
260 Adger, W, *et al*, 2009 *Climatic Change*, 93, 335-354.
261 Reeder, T, *et al*, 2009, *Protecting London from tidal flooding: limits to engineering adaptation*, In: Adapting to Climate Change: Thresholds, Values, Governance, eds, W, Neil Adger, Ireni Lorenzoni, and Karen L. O'Brien, Published by Cambridge University Press 2009.
262 Donovan, B, *et al*, 2006, *Introducing the concept for decision pathways to help make adaptive and robust flood risk management decisions under uncertainty*, Paper presented at the Flood and Coastal Erosion Risk Management Conference, Defra, London.
263 Reeder, T, *et al*, 2009, *Protecting London from tidal flooding: limits to engineering adaptation*. In: Adapting to Climate Change: Thresholds, Values, Governance, eds. W. Neil Adger, Ireni Lorenzoni and Karen L. O'Brien. Published by Cambridge University Press 2009.
264 Colburne, L, Straw, E, 2009, *Evaluation of the use of Working with Others – Building Trust for the Shaldon Flood Risk Project*, Includes cost-benefit analysis of various approaches to community engagement and presents recommendations for organisational change to support genuine engagement, Environment Agency
265 https://consult.environment-agency.gov.uk/portal/ho/flood/fcerm/strategy
266 http://www.defra.gov.uk/corporate/consult/flood-coastal-erosion/index.htm
267 Trench, B, 2009, *Towards an analytical framework of science communication models*, In: Cheng, D. and Claessens, M. and Gascoigne, T. and Metcalfe, J. and Schiele, B. and Shi, S., (eds.) Communicating science in social contexts: new models, new practices. Springer Netherlands, pp. 119-138.
268 Colburne, L, Straw, E, 2009, *Evaluation of the use of Working with Others – Building Trust for the Shaldon Flood Risk Project*, Includes cost-benefit analysis of various approaches to community engagement and presents recommendations for organisational change to support genuine engagement. Environment Agency.
269 Fish, R, 2010, *Participation and an ecosystems approach to decision making*, Draft Guidelines NR0124
http://randd.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=15230
270 Ludwig, D, 2001, *Ecosystems* 4, 758-764.
Horst, R, and Webber,M, *Dilemmas in a General Theory of Planning*, pp. 155–169, Policy Sciences, Vol. 4, Elsevier Scientific Publishing Company, Inc., Amsterdam, 1973.
271 Dobson, A, Bradshaw, A, Baker, A, 1997, *Science* 277, 515–524
Ecosystem Services for Poverty Alleviation conceptual framework for analysis of sustainable management of ecosystems to maximise poverty alleviation - http://www.nerc.ac.uk/research/programmes/espa/documents/Final%20Report%20China%20-%20annex.pdf
272 FAO, 1996, *Rome Declaration on World Food Security and World Food Summit Plan of Action*, World Food Summit, 13-17 November 1996, Rome.
273 Ericksen, P, 2008, *Global Environmental Change*, 18(1): 234 – 245.
274 http://www.bis.gov.uk/assets/bispartners/foresight/docs/food-and-farming/11-546-future-of-food-and-farming-report.pdf
275 Godfray, H, *et al*, 2010, *Science*, 327 (5967), 812-818
276 Smith, P, et al, 2010, *Phil. Trans. R. Soc. B*, 365, 2941-2957
277 Garcia, S, Rosenberg, A, 2010, *Phil. Trans. R. Soc. B*, 365, 2869-2880
278 Bostock, J, et al, 2010, *Phil. Trans. R. Soc. B*, 365, 2897-2912
279 Royal Society of London, 2009, *Reaping the Benefits: Science and the Sustainable Intensification of Global Agriculture*, Royal Society, London
280 World Bank, 2008, *World Development Report, Agriculture for Development*, World Bank, Washington DC
Royal Society of London, 2009, *Reaping the Benefits: Science and the Sustainable Intensification of Global Agriculture*, Royal Society, London
281 Godfray, H, *et al*, 2010, *Phil. Trans. R. Soc. B*, 365, 2769-2777
282 Defra, 2010, *Food Security Assessment: Detailed Analysis*
283 Strzepek, K, Boehlert, B, 2010, *Phil. Trans. R. Soc. B*, 365, 2927-2940
284 Smith, P, *et al*, 2010, *Phil. Trans. R. Soc. B*, 365, 2941-2957
285 C. Nellemann *et al*, Eds, 2009, *The Environmental Food Crisis* (United Nations Environment Programme (UNEP), Nairobi, Kenya
286 Gornall, J, *et al*, 2010, *Phil. Trans. R. Soc. B*, 365, 2973-2989
287 Jaggard, K, Aiming, Q, Ober, E, 2010, *Phil. Trans. R. Soc. B* 365, 2835-2851
288 Woods, J, *et al*, 2010, *Phil. Trans. R. Soc. B*, 365, 2991-3006
289 Piesse, J, Thirtle, C, 2010, Phil. Trans. R. Soc. B, 365, 3035-3047
290 FAO, 2009, *Statistics from www.faostat.fao.org*, updated April 2009, Rome, Italy: FAO.
291 Gallai, N, *et al*, 2009, *Ecology Economics* 59 (3), 287-295
292 Power, A, 2010, *Phil. Trans. R. Soc. B* 365, 2959-2971
293 Jackson, L, Bawa, K, Pascual, U, Perrings, C, 2005, *agroBIODIVERSITY: A new science agenda for biodiversity in support of sustainable agro-ecosystems*, DIVERSITAS Report No 4. 40pp.
294 Power, A, 2010, *Phil. Trans. R. Soc. B*, 365, 2959-2971
295 Maes, W, Heuvelmans, G, Muys, B, 2009, *Environ. Sci. Technol*, 43, 7324-7330.
296 Power, A, 2010, *Phil. Trans. R. Soc. B*, 365, 2959-2971
297 Natural England, 2009, *Food Security and Environment Policy*
298 Power, A, 2010, *Phil. Trans. R. Soc. B*, 365, 2959-2971
299 Thorton, P, 2010, *Phil. Trans. R. Soc. B* 365, 2853-2867.
300 Pretty, J, *et al*, 2006, *Environmental Science and Technology* 40 (4), 1114-1119
301 Godfray, H, 2010, *Phil. Trans. R. Soc. B*, 365, 2769-2777
302 Kearney, J, 2010, *Phil. Trans. R. Soc. B* 365,2793-2807
303 http://www.gecafs.org/
304 Sustainable Development Commission, 2008, *Green, healthy and Fair: a review of the Government's role in supporting sustainable supermarket food*, Sustainable Development Commission, London.
305 Royal Commission on Environmental Pollution, 2007, *The Urban Environment*, Twenty-Sixth Report
The Government Office for Science, London, 2010, *Foresight Land Use Futures Project*, Final Project Report
The Government Office for Science, 2004, *Foresight Flood and Coastal Defence Project*, London
House of Lords Science and Technology Committee, 2006, *Water Management*, Eighth Report of session 2005-2006
306 GECAFS 2010 Presentation

307 Jordan, A, Lenschow, A, 2008, *Integrating the environment for Sustainable Development: An Introduction*, In :Innovation in Environmental Policy: Integrating the Environment for Sustainability, Edited by Jordan, A.J., Lenschow, A. Edward Elgar Publishing Limited.
308 Hertin, J, Jacob, K, Volkery, A, 2008, *Policy Appraisal*, In :Innovation in Environmental Policy: Integrating the Environment for Sustainability, Edited by Jordan, A, Lenschow, A, Edward Elgar Publishing Limited
309Turnpenny, J, *et al*, 2009, *Journal of European Public Policy,* 16(4), 640-653
Juntti, M, Russel, D, Turnpenny, J, 2009, *Environmental Science and Policy* 12, 207-215
Turnpenny, J, Lorenzoni, I, Jones, M, 2009, *Environmental Science and Policy* 12, 347-358
Turnpenny, J, et al, 2008, *Journal of Environmental Planning and Management* 51 (6), 759-775
310 Dobson, A, 2010, SDRN Briefing 5: *Environmental citizenship and pro-environmental behaviour*.
311 Brooks, T, *et al*, 2002, *Conservation Biology*, 16, 909-923
312 Guo, Z, Zhang, L, Yiming, L,2010, *PLoS ONE*, 5(10) c13113
313 Haines-Young, R, Potschin, M, 2009, *Methodologies for defining and assessing ecosystem services*, Final Report, JNCC
Hooper, D, et al, 2005, *Ecological Monographs*, 68,121-149
314 Violle C *et al*, 2007, *Oikos*, 166, 882-892.
315 Hawlins, B, 1999, *Oikos*, 86, 493-506.
Losey, J, Vaughan, M, 2006, *Bioscience* 56, 311-323
316 Diaz, S, *et al*, 2004, *Journal of Vegetation Science* 15(3): 295-304.
317 Anton, C, *et al*, 2010, *Biodiversity Conservation*, 19(10): 2979-2994
318 Lavorel S, *et al*, 2010, *The RUBICODE Project, Rationalising Biodiversity Conservation in Dynamic Ecosystems*, Funded under the European Commission Sixth Framework Programme.
De Deyn, G, Cornelissen, J, Bardgett, R, 2008, *Ecology Letters*, 11, 516-531.
319 Kremen, C, 2005, *Ecological Letters* 8, 468-479.
320 De bello F, 2009, Biodiversity and Conservation , 19 (10), 2873-2893
321 Hegland, S, *et al*, 2009, *Ecological Letters* 12,184-195
322 Flynn, D, et al, 2009, *Ecology Letters* 12, 22-33.
323 Lavorel S, *et al*, 2010, *The RUBICODE Project, Rationalising Biodiversity Conservation in Dynamic Ecosystems*, Funded under the European Commission Sixth Framework Programme.
324 European Academies Science Advisory Council, 2009, *Ecosystem Services and Biodiversity in Europe*, EASAC Policy Report 09
325 Bradley J, *et al*, 2006, *Nature*, 443, 989 -992
Cardinale, B, *et al*, 2007, *PNAS* 104, 18123-18128
326 Gross, N, *et al*, 2009, *Functional Ecology*, 23, 1167-1178
327 Balvanera, P, *et al*, 2006, *Ecology Letters*, 9, 1146–1156
Schmid, B, et al, 2009, *Consequences of species loss for ecosystem functioning: a meta-analysis of data from biodiversity experiments, pages 14-29 in Biodiversity and Human Impacts*, S. Naeem, D. Bunker, M. Loreau, A. Hector, C. Perring. Oxford University Press.
Cardinale, B, et al, 2009, *Ecology* (Data Paper), 90:854.
Power, L, Cardinale, B, 2009, *Oikos*, 118(11), 1703-1711
328 Díaz, S, et al, 2006, *PLOS Biology* 4 (8):1300-1305.
329 Hector, A, Bagchi, R, 2007, *Nature*, 448, 188-190.
330 Haines-Young, R, 2009, *Environmental Limits, Ecosystem Resilience and Supporting Services*, Pilot Review Report for Defra.
331 De bello F, 2009, Biodiversity and Conservation , 19 (10), 2873-2893
332 Catchpole R, 2010, *National Climate Change Vulnerability Assessment*, Natural England Technical Information Note.
333 Fussel, H-M, Klein, R, 2006*, Climate Change*, 75, 301-329.
334 Mitchell, R, *et al*, 2007, *England Biodiversity Strategy – Towards adaptation to climate change*, Final Report to Defra, contract CR0327.
335 Hanski, I, 1999, Metapopulation ecology, Oxford: Oxford University Press
336 Maclean, N, 2010, *Silent Summer: the state of wildlife in Britain and Ireland*, Cambridge University Press, Cambridge
337 Hanski, I. 1999. *Metapopulation ecology*, Oxford, Oxford University Press.
338 Mace, G, et al, 2009, *The biological limits to adaptation: What is known and not known about genetic and phenotypic responses to rapid and extreme environmental change*, Pilot Review for Defra,
339 Carpenter, S, *et al*, 2009, *PNAS*, 106(5),1305-1312
340 Haberl, H, et al, 2005, *Agriculture, Ecosystems and Environment*, 110, 119 – 131.
341 EEA, 2005, *The European Environment – State and Outlook 2005, State of the environment report*, European Environment Agency, Copenhagen
342 EEA, 2010, *Assessing biodiversity in Europe – the 2010 report*, EEA Report No5/2010
343 FAO, 2000a, *Global Forest Resources Assessment 2000, On definitions of forest and forest change*, FRA Working Paper 33, Rome: FAO
344 World Bank, 2007, *At Loggerheads?*
345 UNFCCC, 2007, *Initial analysis on the mitigation potential in the forestry sector*
346 Rudel, T, *et al*, 2005, *Global Environmental Change*, 15 23-31.
347 TEEB, 2010, *The Economics of Ecosystems and Biodiversity for National and International Policy Makers*
348 Betts, R, Sanderson, M, Woodward, S, 2008, *Philos Trans R Soc Lond B Biol Sci*., 363(1498): 1873–1880.
349 Secretariat of the Convention on Biological Diversity, 2010, *Global Biodiversity Outlook 3*, Montreal, 94 pages.
350 UNFCCC, 2007, *Initial analysis on the mitigation potential in the forestry sector*
351 Wall, D, Snelgrove, P, Covich, A, 2001, *Conservation priorities for soil and sediment Invertebrates*, In Soulé, M.E., Orians, G.H. (eds), Conservation Biology, Research Priorities for the Next Decade, Island Press, Society for Conservation Biology, Washington, DC, pp.99-123
352 Turbé, A, et al, 2010, *Soil biodiversity: functions, threats and tools for policymakers*, Bio Intelligence Service, IRD, and NIOO report for the European Commission, Available at: http://ec.europa.eu/environment/soil/pdf/biodiversity_report.pdf.
353 Gardi, C, et al, 2009, *European Journal of Soil Science* 60, 807-819.
354 Lev S. Kuchment, 2004, *The Hydrological Cycle and Human Impact on it, in Water Resources Management*, [Eds. Arjen Y. Hoekstra, and Hubert H.G. Savenije], in Encyclopedia of Life Support Systems (EOLSS), Developed under the Auspices of the UNESCO, Eolss Publishers, Oxford ,UK, [http://www.eolss.net]
355 Cao, S, *et al*, 2010, *Ambio,* 39(4), 279-283.
356 Vörösmarty, C*, et al*, 2010, *Nature*, 467, 555-561.
357 Strzepek, K, Boehlert, B, 2010, *Phil. Trans. R. Soc. B*, 365, 2927-2940
358 Vörösmarty, C, *et al*, 2010, *Nature*, 467, 555-561
Strayer, D, Dudgeon, D, 2010, *J. N. Am. Benthol. Soc*, 29, 344-358.
359 EEA, 2010, *Assessing biodiversity in Europe – the 2010 report*, EEA Report No5/2010

360 Palmer, A, 2010, *Nature*, 467, 534-535
361 Micklin, P, 2007, *Annual Review of Earth and Planetary Sciences*, 35 (4): 47–72
362 Milly, P, *et al*, 2008, *Science*, 319, 573-574.
363 The Government Office for Science, 2004, *Foresight Flood and Coastal Defence Project*, London
364 EEA, 2009, *Water resources across Europe - confronting water scarcity and drought*, Report no2/2009.
365 Source Environment Agency 2007
366 The Government Office for Science, London, 2010, *Foresight Land Use Futures Project*, Final Project Report
367 Halpern, *et al*, 2008, Science, Vol. 319, no. 5865, 948 - 952
368 RCEP. 2004. *Turning the Tide, Addressing the Impact of Fisheries on the Marine Environment*, 25th Report.
369 Failler, P, 2007, *Future prospects for fish and fishery products, Chapter 4: Fish Consumption in the EU in 2015 and 2030,* FAO Fisheries Circular 972/4 FIEP/c972/4, FAO Rome
370 Hiddink *et al*, 2006. *Can. J. Fish Aquat. Sci*. 63, 721-736
371 Harris *et al*, 2007. *Marine Biology*, 151,973-983
372 Secretariat of the Convention on Biological Diversity, 2010, *Global Biodiversity Outlook 3*, Montreal, 94 pages
373 Valiela, Fox, 2008, Science, Vol. 319, 290-291
374 Secretariat of the Convention on Biological Diversity, 2010, *Global Biodiversity Outlook 3*, Montreal, 94 pages
375 EEA, 2010, *Assessing biodiversity in Europe – the 2010 report*, EEA Report No5/2010
OSPAR ,2009, *Marine litter in the North-East Atlantic Region*
376 House of Commons Environmental Audit Select Committee, *Air Quality*, Fifth Report of Session 2009-2010, HC 229-I
National Audit Office. 2010, Air Quality, Briefing for Environmental Audit Committee.
377 House of Commons Environmental Audit Select Committee, *Air Quality*, Fifth Report of Session 2009-2010, HC 229-I
378 IPCC, '*Summary for Policy Makers' In Climate Change 2007: The Physical Science Basis*. Contribution of Working Group 1 to the Fourth Assessment Report of the IPCC. Cambridge University Press, 2007.
379 WMO, 2009, WMO Greenhouse Gas Bulletin, *The State of Greenhouse Gases in the Atmosphere Using Global Observations through 2008*, No5, 23 November 2009, Geneva
380 IPCC 2007, Working group I; conversions using Matthews, 2008
381 http://www.ipcc.ch/
http://webarchive.nationalarchives.gov.uk/+/http://www.hm-treasury.gov.uk/sternreview_index.htm
http://www.theccc.org.uk/home
382 WMO, 2009, WMO Greenhouse Gas Bulletin, *The State of Greenhouse Gases in the Atmosphere Using Global Observations through 2008*, No5, 23 November 2009, Geneva
383 WMO, 2010, WMO statement on the status of global climate in 2009, WMO – No 1 055, World Meteorological Organisation, Geneva
384 National Audit Office, 2009, Adapting to Climate Change
Defra, 2009, Adapting to Climate Change, UK Climate Projections
Defra, 2008. Adapting to Climate Change in England: A Framework for Action
385 EEA, 2010, *Ecosystem Accounting and the Cost of Biodiversity Losses, The Case of Coastal Mediterranean wetlands*, ISSN Technical report series
386 Ekins, P, 2001, *Journal of Environmental Assessment Policy and Management* 3 (1), 61-93
Stigliz, J, Sen, A, Fitoussi, J, 2009, *Report of the Commission on the Measurement of Economic Performance and Social Progress (CMEPSP)*
387 European Commission, 2009, *GDP and Beyond: Measuring Progress in a Changing World*, Communication from the Commission to the Council and the European Parliament, COM (2009) 433 final
388 Chambers, N, et al, (2004) *Scotland's Footprint*, Best Foot Forward.
389 Eaton, R, Hammond, G, Laurie, J, 2007, *Landscape and Urban Planning* 83, 13-28.
390 Fiala, N, (2008), *Ecological Economics* 67 (4), 519–525
391 Kitzes, J, et al, 2009, *Ecological Economics* 68(4), 929-930
392 Ewing B, *et al 2008*, *The Ecological Footprint Atlas 2008*
393 BSI, 2008, *Specification for the assessment of the life cycle greenhouse gas emissions of goods and services*, British Standards, London
394 Van der Voet, E, et al, 2005, *Policy Review on Decoupling: Development of indicators to assess decoupling of economic development and environmental pressure in the EU-25 and AC-3 countries*, CML report 166, Leiden, Institute of environmental sciences (CML), Leiden, Leiden University
395 EEA, 2006, *Land Accounts for Europe 1990-2000, Towards integrated land and ecosystem accounting, EEA Report 11/2006*, Copenhagen.
396 CLC, 2006, *Corine land cover, Corine land cover 2006 raster data*, www.eea.europa.eu/data-and-maps/data/corine-land-cover-2006-raster.
397 Price, R, Durham, C, Chan, J, 2010, *Government Economic Service Review of the Economics of Sustainable Development*, Government Economic Service and Department for Environment, Food and Rural Affairs.
398 Feld, C, et al, 2010, *Biodiversity and Conservation*, 19, 2895-2919
399 EEA, 1999, *Environmental Indicators: typology and overviews*, EEA Tech. Rep. No. 25, Copenhagen.
400 EEA, 2006, *Land accounts for Europe 1990-2000*, EEA report No 11/2006 prepared by Haines-Young, R. And Weber, J-L., EEA, Copenhagen, Denmark. URL: http://reports.eea.europa.eu/eea_report_2006_11/en.
401 EEA, 2010, *Ecosystem Accounting and the Cost of Biodiversity Losses, The case of coastal Mediterranean wetlands*, ISSN Technical report series
402 Haines-Young, R, Potschin, M, 2010, *Proposal for a Common International Classification of Ecosystem Goods and Services (CICES) for Integrated Environmental and Economic Accounting*, Report to the EEA Contract No. EEA/BSS/07/007
403 Feld, C, et al, 2009, *Oikos*, 118 (12), 1862-1871.
404 Eigenbrod, F, et al, 2010, *Journal of Applied Ecology*, 47, 377-385.
405 Linstead, C, *et al*, 2008, *Reviewing Targets and Indicators for the Ecosystem Approach*, Final Report, Defra Project Code NR0119
406 Haines-Young, R, Potschin, M, 2010, *Proposal for a Common International Classification of Ecosystem Goods and Services (CICES) for Integrated Environmental and Economic Accounting*, Report to the EEA Contract No. EEA/BSS/07/007
407 Srivastava, D, Vellend, M, 2005, *Ann Rev Ecol Evol System*, 36, 267-294
408 Layke, C, 2009, *Measuring Nature's Benefits: A preliminary Roadmap for Improving Ecosystem Service Indicators*, Working paper, World Resources Institute Washington DC
409 http://www.esindicators.org/
410 Smart, S, *et al*, 2010, *An Integrated Assessment of Countryside Survey data to investigate Ecosystem Services in Great Britain*, CS Technical Report 10/07, Centre for Ecology and Hydrology
411 http://www.cbd.int/ecosystem/principles.shtml
412 http://www.bis.gov.uk/Policies/better-regulation/policy/scrutinising-new-regulations/preparing-impact-assessments
413 http://www.bis.gov.uk/assets/biscore/better-regulation/docs/10-901-impact-assessment-toolkit.pdf

414 Price, R, Durham, C, Chan, J, 2010, *Government Economic Service Review of the Economics of Sustainable Development*, Government Economic Service and Department for Environment, Food and Rural Affairs.
415 http://www.defra.gov.uk/corporate/policy/guidance/sd-impact/index.htm
416 CLG, 2010, *Decentralisation and the Localism Bill: an essential guide*.
House of Commons Library, 2011, *Localism Bill: Planning and Housing*, Research Paper 11/03
417 http://www.scotland.gov.uk/Publications/2010/09/23100015/0
418 Land Use Consultants, 2006, *A practical toolkit for assessing cumulative effects of spatial plans and development projects on biodiversity in England*, English Nature Research Reports, No 673: http://www.english-nature.org.uk/pubs/publication/PDF/673.pdf
419 ENTEC, 2010, CTX 0811, *Review of the Biodiversity Duty contained in Section 40 of the NERC Act 2006*, Report to Defra. http://randd.defra.gov.uk/Document.aspx?Document=WC0788_9135_FRP.pdf
420 GONW, 2008, *North West of England Plan – Regional Spatial Strategy to 2021*, London, TSO
421 NENW, 2008, *Developing an Outline Strategy for Linking Grey and Green Infrastructure*, Kendal
422 Biodiversity South West, 2007, *South West Nature Map- A Planners Guide, Helping to Shape Spatial Planning for Biodiversity in Local Development Frameworks*
423 Ashworth, P, et al, 2008, *Case Study to Develop Tools and Methodologies to Deliver an Ecosystems Approach – Heysham to M6 Link,* ADAS, Defra Research Project NR0110
424 DECC, 2010, *Severn Tidal Power SEA Environmental Report*, Prepared by Parsons-Brinckerhoff Ltd
425 Presentation Ozdemiroglu, E, Eftec, CIWEM-CMS Conference *Valuing Water and the Environment in Practice*, 2010. Available at http://www.ciwem.org/media/182352/Combining%20Ecosystem%20Services%20and%20Economic%20Valuation%20for%20Policy%20and%20Project%20Appraisal.pdf
426 http://www.grasslands-trust.org/campaign.php?campaignid=4
427 Such as http://www.bailii.org/ew/cases/EWCA/Civ/2010/608.html
428 Byron, H, *et al*, 2000, *Journal of Environmental Planning and Management*, 43(1), 71–97
Latimer, W, 2009, *Planning Practice and Research*, 24, 3, 389-408
429 Ashworth, P, et al, 2008, *Case Study to Develop Tools and Methodologies to Deliver an Ecosystems Approach – Heysham to M6 Link,* ADAS, Defra Research Project NR0110
430 RICs, 2007, *Guidance Note: Environmental Impact Assessment*, 2007 RICS Books
431 Ashworth, P, et al, 2008, *Case Study to Develop Tools and Methodologies to Deliver an Ecosystems Approach – Heysham to M6 Link,* ADAS, Defra Research Project NR0110
432 Latimer, W, 2009, *Planning Practice and Research*, 24, 3, 389-408
433 Hill, D, 2009, *Regulation of Standards in Environmental Mitigation Associated with Development*, In Practice, IEEM, Winchester.
434 Treweek Environmental Consultants, 2009, *Scoping study for the design and use of biodiversity offsets in an English Context*, Report to Defra
435 Treweek, J, 1999, *Ecological Impact Assessment*, Blackwell Science, Oxford

Abbrevations

AES	Agri-environment Scheme
BAP	Biodiversity Action Plan
BIS	Department of Business, Innovation and Skills
CAP	Common Agricultural Policy
CAS	Complex Adaptive System
CBA	Cost Benefit Analysis
CBD	Convention on Biological Diversity
CICES	Common International Classification for Ecosystem Services
CLG	Department of Local Government and Communities
CMEPSP	International Commission on the Measurement of Economic Performance and Social Progress
CORINE	Co-ordination of Information on the Environment
Defra	Department for Environment, Food and Rural Affairs
DSPIR	Driving Force-Pressure-State-Impact-Response
EAC	House of Commons Environmental Audit Select Committee
EASAC	The European Academies Science Advisory Council
EcIA	Ecological Impact Assessment
EEA	European Environment Agency
EIA	Environment Impact Assessment
EMC	Environmentally weighted Material Consumption
EPI	Environmental Policy Integration
ERZ	Ecosystem Restoration Zone
ES	Environmental Stewardship
ESP	Ecosystem Service Providers
ESRC	Economic and Social Research Council
ESV	Ecosystem Service Valuation
ESVA	Ecosystem Service Valuation Assessment
EU	European Union
FAO	Food and Agriculture Organisation
FCERM	Flood and Coastal Erosion Risk Management
FCSG	Full Costs of Goods and Services
GDP	Gross Domestic Product
GHGs	Greenhouse Gas Emissions
GIS	Geographic Information System
GLOBE	International Commission on Land Use Change and Ecosystems
GNP	Gross National Product
GPP	Gross Primary Production
GQA	General Quality Assessment Exercise
HANPP	Human Appropriation of Net Primary Productivity
HLS	Higher Level Stewardship
IA	Impact Assessment
IDP	Inclusive Domestic Product
IPBES	Intergovernmental Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
JNCC	Joint Nature Conservation Committee
LCA	Life Cycle Analysis
LEAC	Land Cover and Ecosystem Accounts
LWEC	Living With Environmental Change research programme
MA	Millennium Ecosystem Diversity

MCZ	Marine Conservation Zone
MFA	Materials Flow Analysis
NAMEA	National Accounting Matrices Including Environmental Accounts
NAO	National Audit Office
NDP	National Domestic Product
NEA	National Ecosystem Assessment
NERC	Natural Environment Research Council
NNR	National Nature Reserve
NPP	Net Primary Production
NPS	National Planning Statement
NRES	Natural Resources and related ecosystem services
OECD	Organisation for Economic Co-operation and Development
OFWAT	The Water Services Regulation Authority
PES	Payment for Ecosystem Services
PPS	Planning Policy Statement
REDD	Reducing Emissions from Deforestation and Degradation
RELU	Rural Economy and Land Use research programme
RIA	Regulatory Impact Assessment
ROAMEF	Rationale, Objectives, Appraisal, Monitoring, Evaluation and Feedback
RSPB	Royal Society for the Protection of Birds
RUBICODE	Rationalising Biodiversity Conservation in Dynamic Ecosystems Project
SA	Sustainability Appraisal
SAC	Special Area for Conservation
SCaMP	Sustainable Catchment Management Plan
SEA	Strategic Environmental Assessment
SEEA	The UN System of Environmental and Economic Accounts
SES	Social Ecological Systems
SGAP	Sustainability Gap
SMS	Safe Minimum Standards
SNA	Strategic Nature Area
SPA	Special Protected Area
SPU	Service Providing Unit
SSSI	Sites of Special Scientific Interest
TE2100	Thames Estuary 2100 project
TEEB	The Economics of Ecosystems and Biodiversity
TEV	Total Economic Value
UNEP	United Nations Environment Programme
UNFCCC	United Nations Convention on Climate Change
WFD	Water Framework Directive
WTA	Willingness to Accept
WTP	Willingness to Pay
WWF	World Wildlife Fund

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