03/05/2013

Biotic and Biophysical Underpinning of Ecosystem Services in the Scottish Context: A Review

Rob Brooker[1], Inge Aalders[1], Keith Ballingall[2], Graham Begg[1], A. Nick E. Birch[1], Geoff Elliott[1], Chris Ellis[3], Thomas Freitag[1], Cathy Hawes[1], John Holland[4], Bex Holmes[1], Rupert Hough[1], Alison Karley[1], Davy McCracken[4], Ruth Mitchell[1], Jenni Stockan[1], Ruth Zadoks[2], Steve Albon[1], Alison Hester[1], Robin Pakeman[1]

1. The James Hutton Institute

2. The Moredun Research Institute

3. The Royal Botanic Gardens Edinburgh

4. SRUC: Scotland's Rural College

Contents

Executive Summary

This review has been undertaken as part of the Ecosystem Services Theme of the Scottish Government Strategic Research Programme: Environmental Change.

The aim of this review is to help deliver the request from Scottish Government for:

> *Increased understanding of the linkages between the primary ecological and evolutionary processes, ecosystem function and ecosystem services, to inform assessment of the consequences of environmental change for the wide range of ecosystem services. (RD 1.1.2).*

By undertaking a review exercise focussed on the underpinning of ecosystem service delivery by the natural environment we are able to:

1. Improve shared understanding across the Work Packages, Themes, and Programmes about ecosystem service delivery and the Ecosystem Approach concepts.

2. Better target future research activity toward identified knowledge gaps.

This is a rapidly-developing research field, and the breadth of topics and information that might be covered by such a review is very large. Consequently, the review is focussed around topics which are relevant to the needs of Scottish Government and which will inform future research activity by the Ecosystem Services Theme of the RESAS Strategic Research Programme.

Our ***Introduction (Chapter 1)*** sets out the wider context for the work. It explains the origin of the Ecosystem Approach and ecosystem service concepts. It looks briefly at the wide range of research being conducted globally on biodiversity-ecosystem service linkages. It explains the approach that we have taken to focus the review to make it more policy-relevant within a Scottish context.

At the heart of this approach is an alignment with 'broad policy goals' – around which chapters 2 through to 5 are structured - as well as the identification of prioritised ecosystem services. The prioritisation of services took place as part of the second Ecosystem Approach Working Group workshop, and was based upon expert judgement and opinion from participating stakeholders. The prioritisation process is explained in more detail in Chapter 1. The broad policy goal chapters consider how biodiversity and biotic/biophysical processes underpin the delivery of prioritised services.

For clarity, Chapter 1 also provides our working definitions for key concepts and terminology which are used throughout.

With respect to a ***Low carbon economy (Chapter 2)*** trees, peat, soil formation and crops were considered by the workshop participants to be the most important ecosystem services. It is the carbon sequestration provided by these that is particularly important, contributing to the final ecosystem service of climate regulation.

- With respect to ***trees***, biophysical conditions influence species growth and carbon storage but may be overridden by management practices such as fertilisation and drainage. Biotic processes both enhance and restrict the ecosystem service of trees, through mycorrhizal associations, pollination, dispersal of seeds/fruits, pest regulation, disease, and browsing by herbivores. Increased tree cover results in more vegetation-stored carbon, but this is balanced against carbon loss from soils in some instances. Abiotic and biotic factors and forestry management influence total carbon storage by trees.
- With respect to ***peat and soil formation***, extraction of peat leads to GHG emissions, in conflict with the 'good' of an equitable climate. Peat formation is underpinned by biotic processes of peat-forming plants and associated litter-decomposing microbial communities.

Sphagnum-dominated bryophyte communities are the main peat builders in high latitudes. While Sphagnum has been described as an 'ecosystem engineer', reflecting its importance in peatland formation, the impact of associated microbial decomposer biodiversity on peat formation is unclear. Most soil processes, including soil formation, are mediated by the soil microbial community, which in turn is strongly influenced by plant community structure. Although the huge biological diversity found in soils may appear relatively 'inactive', it may be central to system or functional resilience; therefore discounting this diversity may fail to account for a key regulating component of the soil environment and its processes.

- With respect to ***crops***, bioethanol, biodiesel, food and fibre from reduced input farming are the 'goods' of greatest relevance. Crops play a key role in delivering a low carbon economy because land use or crop types might be directly targeted to support renewable energy actions, and changes in crop management also impact on carbon emissions. The impact of biodiversity on crop production is often positive (e.g. pollinators), but biodiversity effects are relatively small compared to management actions as well as geographic and temporal variation in soil conditions.
- ***Overall***, upland habitats generally provide more carbon storage (trees and peat) than lowland habitats, while lowland habitats generally provide more food and fibre, but these broad generalizations hide much local variation. The most common conflicts or trade-offs occur around land use and land management; for example, decisions are required on how best to manage land as a limited resource in providing different crops (food or biofuels) or protected habitats, and ecosystem service mapping is invaluable in this respect. However, there are gaps in our understanding of how biodiversity and biotic/biophysical processes underpin the delivery of ecosystem services relevant to a low carbon economy.

With respect to ***sustaining food production (Chapter 3)***, crops, livestock, soil formation, and pollination were considered by workshop participants to be the most important ecosystem services.

- With respect to ***crops***, Scottish production is highly mechanised, with considerable inputs and high intervention. Consequently, although biophysical conditions are a major determinant, biodiversity currently has a limited underpinning role. However, many studies find positive relationships between biodiversity and relevant functions (e.g. productivity or pest and disease regulation), often thought to result from the characteristics of the species concerned and their impacts on ecosystem function. Declining biodiversity could have consequences for ecosystem functions central to crop production, and ultimately for its productivity and sustainability.
- ***Livestock*** production is the dominant agricultural sector in Scotland, and is particularly important for the uplands. There is a considerable literature on how livestock production affects biodiversity and biophysical processes (both positively and negatively), but rather little information on its underpinning by biodiversity. However, biophysical drivers and processes (e.g. climate, soil and water conditions) clearly can impact livestock directly by influencing grazing quality, and indirectly through regulating disease and pests.
- For ***Pollination*** services, the link between biodiversity and pollination is strong and clear: a reduction in pollinators can be expected to have a deleterious effect on this ecosystem service. However, a relatively small proportion of current Scottish crop production is dependent on pollination (about 13% of total output value). In addition, since wind-pollinated grasses are the main source of fodder, there is likely to be no impact of pollinator losses on the production of meat and dairy products or on grain production.
- ***Soil formation*** is vital for food production. The links between soil formation, biodiversity and biotic and biophysical processes are complex. Research on the role of soil biodiversity in ecosystem function has lagged behind corresponding research above-ground, but functional trait approaches may be useful in addressing research gaps. Soil biodiversity in many areas is

clearly in decline and, as soil biota are a component of healthy soils, the impacts of current farming practice on soil biota may negatively impact on soil formation.

- ***Overall*** it appears that the functional diversity of organisms may be central to sustaining food production. Natural processes, and biodiversity at a range of scales, can help to deliver services directly relevant to sustaining food production, and can do so in a sustainable manner. However, it is clear that we need a better understanding of how to integrate nature conservation with food production, and to balance the negative (e.g. pests and diseases) as well as the positive effects of biotic processes and biodiversity.

With respect to ***halting biodiversity loss (Chapter 4)***, the ecosystem services prioritised by workshop participants for consideration here are wild species diversity (as both a cultural service and provisioning service), disease and pest regulation, and crops.

- ***Wild species diversity (cultural service)*** is more likely to be directly regulated by natural biophysical and biotic processes in upland than in lowland systems. Defining the 'goods' delivered is complex but important: these may differ substantially between stakeholders, and so too might the service's relationship to biodiversity and biophysical processes. Understanding the basis for conservation targets – the "appropriate" level of biodiversity - is also important: this will differ between the lowlands and uplands. All types of biodiversity are likely to play a role in regulating this service
- For ***wild species diversity (provisioning service)***, increased biodiversity is likely to be important for ecological restoration, but this positive relationship is probably weaker for other types of bioprospecting (e.g. the hunt for pharmaceutical products). Increased diversity overall can be beneficial for the provision of harvestable species, with the exception of some particular species groups (epidemic pests and diseases).
- The relationships between biodiversity and biotic/biophysical processes and ***disease and pest regulation*** are complex, not least because either side of the pathogen/pest–host relationship may be affected. We have some knowledge of these relationships from crop and livestock production systems, but our knowledge is poorer for more complex natural and semi-natural systems (although critical with respect to halting biodiversity loss). There is now considerable potential for extending techniques developed in production systems to explore these relationships in natural/semi-natural systems.
- With respect to ***crops***, intensification clearly leads to negative biodiversity impacts. Sustainable farming practices will be beneficial for biodiversity in crop production systems, but the extent to which biodiversity-supported functions can offset the loss for crop production from less intensive farming practice is unclear. Other changes in crop production systems, beyond simply reducing the intensity of management, may have beneficial impacts for farmland biodiversity and can contribute to biodiversity conservation.
- ***Overall***, in all systems it is important to understand which elements of biodiversity are critical for delivering the aims of the broad policy goal, and how these relate to the desires of and management by different stakeholder groups. This level of detail is necessary for developing integrated management practices that promote biodiversity conservation.

Prioritised ecosystem services selected for consideration for ***sustainable water management (Chapter 5)*** are water cycling, water detoxification and purification, and water supply. To deal with the close interconnectedness of these services, Chapter 5 focuses on water quantity and quality: delivery of both involves elements of all three prioritised ecosystem services.

- With respect to ***water quantity***, climate, topography, geology and physical processes play a very substantial role in determining quantity. Perhaps the most critical aspect of biological processes is the occurrence of specific habitats and ecosystems rather than biodiversity *per* *se*. Within these habitats certain groups of organisms, in particular vascular plants and bryophytes, have the biggest impact on water quantity. However, other groups such as soil fungi may have substantial yet currently-unquantified roles. Native ecosystems (semi-natural habitats) tend to have a greater beneficial impact on water quantity compared to those comprised of or dominated by non-native organisms.

- There is a much greater relative role for biological processes in regulating ***water quality***. Land management can determine the functioning of biophysical processes that regulate water quality, e.g. water penetration. As with water quantity, the physical process of water penetration (prior to detoxification) may be dependent on the occurrence of specific ecosystem types, although biodiversity *per se* may also be important in enabling a wide range of potential pollutants to be detoxified. Although there is less certainty about this biodiversity role, different habitats deliver different components of the water cycle that enhance water quality, and new pollutants indicate the potential for apparently 'redundant' components of biodiversity to be of future use in detoxification processes.
- ***Overall*** the uplands are central to delivering both water quantity and quality. Purification processes – enhancing quality - are also important in lowland ecosystems, but there is much greater dependency of lowland users on upland systems than *vice versa*. The dependency between upland and lowland systems is probably much greater than for other ecosystem services. The scale needed for appropriate planning for the delivery of sustainable water management is likely to be much larger (e.g. across entire catchments) compared to the delivery of services important for other broad policy goals.

In our ***Discussion and conclusions (Chapter 6)*** we assess the approach taken for our review, overarching patterns in the relationships between biophysical/biotic processes and biodiversity and ecosystem service delivery, and knowledge gaps.

In terms of our ***approach*** we conclude that:

- The focus on prioritised ecosystem services for practical reasons has not limited the types of services or levels of biodiversity considered;
- Consistent use of terminology is essential, as is the provision of clear definitions for key concepts (such as those used in Chapter 1);
- This review should be seen as part of a process of on-going dialogue and discussion which is helping to deliver improved and shared understanding.

In terms of the ***relationships*** between biodiversity and biotic/biophysical processes and ecosystem services, we conclude that:

- Although biotic and biophysical processes clearly underpin the vast majority of ecosystem services, the role of biodiversity *per se within* an ecosystem is unclear: in many cases it is the occurrence of particular species, functional groups or habitats that seems critical for service delivery, and the diversity of these components *among* locales which is required to sustain Scotland's multifunctional landscape;
- In some cases service delivery is strongly and directly regulated by the physical environment; in others it is mediated by interactions between biotic and physical processes;
- Differences in the physical properties of upland and lowland systems have profound implications for the potential uncoupling of service delivery from any biodiversity/biophysical underpinning. A simple model (below) can be put forward to clarify these relationships. This suggests that indicators that can genuinely monitor the delivery of services will be more effective in monitoring system health in upland rather than in lowland environments in Scotland.

Finally, one of our primary aims was to identify ***knowledge gaps***. Each chapter identifies knowledge gaps that are of particular relevance to that broad policy goal, and some of these have been mentioned above.

In addition, some knowledge gaps are common across broad policy goals, specifically:

- Framing cultural service concepts to explore their underpinning by biodiversity and biotic/biophysical processes;
- Understanding the role of genetic diversity in maintaining ecosystem function and service delivery;
- Understanding the role of functional diversity and species redundancy;
- Understanding the importance of the spatial configuration of habitats/ecosystems, including the possible occurrence of scale-dependent thresholds of function;
- Understanding whether the Ecosystem Approach *will or will not* further enable biodiversity conservation.

This list of generic knowledge gaps should in no way be taken as indicating some form of priority order. In addition, although some knowledge gaps might be considered generic across broad policy goals, this does not mean that they are necessarily of greater importance than those related to particular policy goals. In order to genuinely enhance the application of the Ecosystem Approach and uptake of the ecosystem service concept, it will be necessary to address all of these knowledge gaps.

1. Introduction

1.1 Summary

The aim of this review is to help deliver the request from Scottish Government for:

> *Increased understanding of the linkages between the primary ecological and evolutionary processes, ecosystem function and ecosystem services, to inform assessment of the consequences of environmental change for the wide range of ecosystem services. (RD 1.1.2).*

By undertaking a review exercise focussed on the underpinning of ecosystem service delivery by the natural environment we are able to:

1. Improve shared understanding across the Work Packages, Themes, and Programmes about this aspect of the ecosystem service and Ecosystem Approach concepts,

2. Better target future research activity toward identified knowledge gaps.

This is a rapidly-developing research field, and the breadth of topics and information that might be covered by such a review is very large. It has therefore been helpful to focus the review to make it directly relevant to the needs of Scottish Government and therefore the future research activity by the Ecosystem Services Theme of the RESAS Strategic Research Programme.

This chapter sets out the wider context for the work. It explains the origin of the Ecosystem Approach and ecosystem service concepts. It looks briefly at the wide range of research being conducted globally on biodiversity-ecosystem service linkages. It then explains the approach that we have taken to focus the review to make it more policy-relevant within a Scottish context. Along with information on our approach to undertaking this review, the structure of this review document is explained in more detail.

Finally, it is clear that many of us continue to wrestle with terminology. This is unsurprising when we are working within new multi-disciplinary topic areas, across land use sectors, in a fast-moving research field. In order to make progress with this review it has been necessary for us to try to agree and then work to a common terminology. The final aim of this introductory chapter is to set out our working definitions for key concepts and terminology which are used throughout this review, which might then be taken up and developed for wider use, for example in the Ecosystem Services Theme or across other RESAS Themes.

1.2 The origin and uptake of the Ecosystem Approach and Ecosystem Service concepts

The origin and development of the Ecosystem Approach and associated ecosystem service framework is complex, not least because the field is developing rapidly and its emerging concepts have been and remain fluid. However, here we try to summarise some of the key steps both in their increasing international prominence and within the UK. A very useful summary of the origin of the Ecosystem Approach is given by Haines-Young & Potschin (2007) and developed subsequently (see Haines-Young & Potschin 2010, Howard *et al.* 2011). Many of the key points within this section have been taken from their overview.

1.2.1 The Ecosystem Approach within the Convention on Biological Diversity

The initial impetus for the uptake of a new approach to biodiversity and natural resource management came about in the late 1980s and early 1990s, with the recognition of the limitations of traditional approaches. More recent and wider interest was instigated by the 1995 adoption by the *Convention on Biological Diversity* (CBD) for the Ecosystem Approach as its primary framework for action (Secretariat of the Convention on Biological Diversity, 2000).

Within the CBD the Ecosystem Approach is constituted of a set of 12 principles, adopted by the *Convention of the Parties to the CBD at its Fifth Meeting, Nairobi, 15-26 May 2000* (Decision V/6, Annex 1. CBD COP-5 Decision 6 UNEP/CBD/COP/5/23). Critical elements of the general approach embodied by these principles are:

- an integrated approach to natural resource management;
- management decisions made with a full appreciation of the economic and social context;
- management decisions made within the constraints of, and with understanding of, the biophysical limits of a system;
- an adaptive approach to natural resource management

For a full list of the 12 principles, see Appendix 1. As noted in the more recent UK National Ecosystem Assessment (UK NEA), the Ecosystem Approach is “much more than accepting ecosystems as the core of environmental management. It recognises that people and society are integral components of ecosystems and their management and conservation” (UK NEA Synthesis of the Key Findings).

Haines-Young & Potschin (2007) highlight ongoing discussion of these principles, and that they should not be seen as fixed or final points in the process. They also point out some interesting variation in the terminology associated with this topic, with reference being made in some studies to the “ecosystems approach”, or an “ecosystem-based approach”. Perhaps the most widely used alternative nomenclature is “ecosystem services approach” which – according to Haines-Young and Potschin - emphasises a focus on the output of ‘goods’ and services rather than the broader additional goals of the Ecosystem Approach. Ecosystem services are “the outputs of ecosystems from which people derive benefits” (UK NEA Ch. 2, after MEA 2005).

Haines-Young & Potschin (2007), and references therein, conclude that there is likely to be little merit in trying to be overly-prescriptive about defining what constitutes an/the ecosystem approach, but that instead we should focus on “the contrast between this broad framework, and approaches to environmental management and policy that do not take ecosystems and people into account in an integrated way”. In particular “one of the merits of the Ecosystem Approach is that it helps focus decision makers on longer-term perspectives rather than on shorter-term fixes that may be difficult to sustain.” However, ecosystem services are mentioned in only one of the principles of the Ecosystem Approach as set out in the CBD. The use of “ecosystem services approach” is therefore potentially confusing as it conflates the EA with one of its principles. In this review we discuss only “Ecosystem Approach” and “ecosystem service(s)”, and avoid use of these other terms.

1.2.2 The Millennium Ecosystem Assessment

A second critical step in policy uptake of the Ecosystem Approach was the *Millennium Ecosystem Assessment* (the MEA; also referred to as the MA). Secretary General Kofi Annan’s 2000 report to the UN (www.un.org/millennium/sg/report/full.htm) called for the development of the MEA, the objective of which was:

> *to assess the consequences of ecosystem change for human well-being and to establish the scientific basis for actions needed to enhance the conservation and sustainable use of ecosystems and their contributions to human well-being.* (MEA 2005)

In terms of conceptual development, the MEA took the step of categorising services into the four now-common groupings of supporting, regulating, provisioning and cultural services. The MEA reported in 2005, concluding first that over the preceding 50 years there has been a substantial and largely irreversible loss in the diversity of life on Earth. Second, although there have also been substantial net gains in human well-being and economic development, these gains are unsustainable. Third, the degradation of ecosystem services could grow significantly worse during the first half of this century. Fourth, reversing degradation - while meeting increasing demands for services - will involve significant changes in policies, institutions and practices that are not currently under way. Despite this pessimistic outlook, the MEA also concluded that there was at least some hope if we implemented particular interventions, many of which were central elements of an Ecosystem Approach to natural resource management and decision making.

1.2.3 The UK National Ecosystem Assessment

At the UK level, the UK NEA was a direct response to publication of the MEA. The House of Commons Environmental Audit Committee's (2007) report on the findings of the MA stated that "ultimately the Government should conduct a full MA-style assessment for the UK to enable the identification and development of effective policy responses to ecosystem service degradation".

The UK NEA was initiated in 2009 and completed in 2011. The key findings of the UK NEA have been comprehensively summarised its Synthesis of the Key Findings (http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx). This synthesis lists a large number of knowledge gaps, some of which are clearly relevant to this review, including:

> *The need to refine our understanding of the fundamental processes underpinning ecosystem service delivery by extending observations and experimental manipulations, and improving models of the key mechanisms.*
>
> *That we are unable to fully quantify relationships between UK biodiversity and the ecosystem services they support, and this is particularly true for particular organism groups, including cryptic organisms such as soil microbes.*

The follow-on phase for the UK NEA is now underway (http://uknea.unep-wcmc.org/NEWFollowonPhase/tabid/123/Default.aspx). The aim of the follow-on phase is "to further develop and communicate the evidence base of the UK NEA and make it relevant to decision and policy making at different spatial scales across the UK". It focuses in particular on four areas: economic analysis; cultural ecosystem services and cultural, shared and plural values; analysis of future ecosystem changes; tools and other supporting materials.

1.2.4 Other drivers for the uptake of the Ecosystem Approach into policy

The Ecosystem Approach and ecosystem service concepts are being rapidly taken up in many areas of policy at both a UK and Scottish level. The UK NEA, and before it the MEA, are not the only drivers of this trend. Another key factor has been the lack of success of the 2010 biodiversity targets. In April 2002, the Parties to the Convention on the CBD agreed the global target "to achieve by 2010 a significant reduction of the current rate of biodiversity loss" (Decision VI/26). This was followed by a commitment at the UN World Summit on Sustainable Development later that year to achieve "by 2010 a significant reduction in the current rate of loss of biological diversity." However, despite these very substantial commitments, even before the publication of Global Biodiversity Outlook 3 (Secretariat of the Convention on Biological Diversity 2010) - which assessed success against these targets and concluded that they would not be met - it became clear that there was a continuing and accelerating decline in the status of the global biodiversity resource.

One critical reason for this failure was highlighted in a European Environment Agency study of Environmental Policy Integration (EPI) in EU member states (EEA 2005). EPI should ensure that environmental issues are reflected in policy making across sectors. The EEA assessment concluded that EPI fails because biodiversity conservation has to compete in the policy arena with other sectors such as economic and social development, and that biodiversity conservation is perceived as placing a cost on these other sectors. There may be a hope – certainly in the conservation sector - that by implementing the ecosystem approach, and valuing associated ecosystem services, this perception of cost will be replaced by a perception of benefit, and that biodiversity conservation will be promoted in policy decisions across sectors.

This hope may be reflected in the development of the 2010 biodiversity target's successors: the Aichi targets. In decision X/2, the tenth meeting of the Conference of the Parties to the CBD, (18-29 October 2010, Nagoya, Aichi Prefecture, Japan) adopted a revised and updated Strategic Plan for Biodiversity, including the Aichi Biodiversity Targets, for the 2011-2020 period. This new plan provides the overarching framework on biodiversity, not only for the biodiversity-related conventions, but for the entire United Nations system. The tenth meeting of the Conference of the Parties also agreed to translate this overarching international framework into national biodiversity strategies and action plans within two years.

At the heart of the Aichi targets is the adoption of an ecosystem approach. The rationale for the Strategic Plan begins: "Biological diversity underpins ecosystem functioning and the provision of ecosystem services essential for human well-being", whilst the vision of the Strategic Plan is a world of "Living in harmony with nature" where "By 2050, biodiversity is valued, conserved, restored and wisely used, maintaining ecosystem services, sustaining a healthy planet and delivering benefits essential for all people". The strategic goals of the Plan are:

A. Address the underlying causes of biodiversity loss by mainstreaming biodiversity across government and society

B. Reduce the direct pressures on biodiversity and promote sustainable use

C. Improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity

D. Enhance the benefits to all from biodiversity and ecosystem services

E. Enhance implementation through participatory planning, knowledge management and capacity-building.

Within the Strategic Plan, there is an underlying assumption that biodiversity is essential to the delivery of ecosystem services and that adoption of the ecosystem service and Ecosystem Approach concepts will promote the conservation of biodiversity. National scale biodiversity strategies are now being revised in response to these new targets, and so also take on board the Ecosystem Approach and ecosystem services concepts. It seems reasonable to note, though, that whether or not biodiversity conservation will on balance be promoted by the adoption of the Ecosystem Approach remains an open question. Central to achieving this will be the adoption of more comprehensive and accepted valuation methods, as recognised in Principle 4 of the CBD Ecosystem Approach, and currently being pursued through initiatives such as the UK NEA follow-on phase, The Economics of Ecosystems and Biodiversity (TEEB; http://www.teebtest.org/) and the Valuing Nature Network (http://www.valuing-nature.net/).

1.3 Biodiversity, biophysical processes, and ecosystem function and services – a brief overview

1.3.1 Definitions of biodiversity and biophysical processes

Biodiversity

Although our working definitions of key concepts are set out Section 1.5, it is necessary here to define more clearly what we mean by biodiversity and biophysical processes before going on to consider current general debate concerning how they regulate and underpin ecosystem services.

Article 2 of the CBD provides a widely-used definition of biodiversity:

> '*Biological diversity' means the variability among living organisms from all sources including,* inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of*

which they are part; this includes diversity within species, between species and of ecosystems.

Biodiversity within the CBD is not therefore an inventory of biological entities (a count of different genes, species, or ecosystems) but represents the variability within and among these things.

The UK NEA does not follow the CBD definition. As argued in Ch. 4 of the UK NEA, diversity *per se* "may have only a limited effect on specific ecosystem services" (see below for more information on this point). Instead the UK NEA adopts a pragmatic approach. As biodiversity trend data in the UK are collected at the level of particular taxonomic groups (e.g. Fish, Amphibians, Reptiles, Birds; see UK NEA Table 4.1, p.68, for a full list), the UK NEA considers the role of these groups in delivering ecosystem services.

Some uses of biodiversity move even further from the CBD definition. In nature conservation legislation, for example the *Scottish Biodiversity Strategy* (Scottish Executive 2004), biodiversity is often used as a synonym for nature or the natural environment. From the perspective of conservation agencies the question "how does biodiversity underpin the delivery of ecosystem services" might in some cases by rephrased as "how do natural systems underpin the delivery of ecosystem services", irrespective of the diversity of organisms within those systems.

Clearly, defining biodiversity can be complex and context-specific, balancing purist and pragmatic arguments related to different approaches to definition. Mace *et al.* (2012) summarise the "multi-layered relationship" between biodiversity and ecosystem services, noting that biodiversity can appear at all levels across the ecosystem service hierarchy: "as a regulator of underpinning ecosystem processes, as a final ecosystem service and as a 'good' that is subject to valuation, whether economic or otherwise". Consequently, it is difficult to find a definition for biodiversity that encompasses all of its context-specific applications, but which is not so vague that it ceases to be of use. In this review, therefore, we have used the following terminology:

- 'Biodiversity' is reserved strictly for discussing the diversity of biota at whatever level in the genes-to-habitats hierarchy. When using the phrase we have tried to specify the level of diversity being discussed (genetic, species, functional group, habitat).
- When discussing simply nature, biota, or biotic/natural processes, rather than something that is specifically about *diversity*, we use these other phrases instead.

By using this distinction we hope we have made it clear where the information provided explains how natural systems or biotic processes underpin ecosystem services, and where biodiversity is genuinely relevant to service delivery.

<u>Biophysical processes</u>

Biophysics is the science of the application of the laws of physics to biological phenomena. In an ecological context the term has a slightly different meaning, and in this review we take the biophysical environment as being the combined biotic and abiotic environment of an organism. We are concerned with the roles of biotic (including biodiversity) and biophysical processes in regulating ecosystem services. Because we discuss separately the role of biodiversity and biotic processes, the biophysical environment is then taken as encompassing the elements of the environment beyond biodiversity and biotic processes. Hence we take biophysical processes as including other potential environmental regulators of ecosystem service delivery such as non-organismal soil processes (although we recognise that it may be impossible in some cases to draw an absolute dividing line between these groupings, as some processes may combine both biotic and abiotic elements). In addition we try to make a distinction between biophysical processes and land form. Land form is an element of the natural environment that can strongly influence service delivery, but which is itself not a process.

1.3.2 – Recent biodiversity-ecosystem service research

Throughout many reviews and position papers runs a core question: how do biodiversity, or natural systems or processes, underpin ecosystem service delivery – indeed, do they?

This question is critical in determining whether or how a focus on the delivery of ecosystem services will in turn lead to the protection or further degradation of the natural environment and biodiversity. However, this question is not new. A considerable body of ecological research has been devoted to understanding whether biodiversity in particular has a function, or is simply the consequence of chance evolutionary processes. A number of recent reviews and projects have attempted to synthesise this literature. Overall, they emphasise the complexity of this relationship, including its scale-dependency and context-specificity.

For example, Balvanera *et al.*'s (2006) meta-analysis of experimental studies found that biodiversity effects on ecosystem functioning and services are weaker at the ecosystem level, if biodiversity manipulations are less well-controlled, and with increasing trophic distance between the manipulated and monitored ecosystem elements. Constanza *et al.* (2007) found that that the link between biodiversity and function (net primary productivity – NPP) depended on the temperature context, with negative, neutral or positive correlations in low-, mid- and high temperature systems, respectively.

During attempts to unpick this complexity, two distinct research areas have recently emerged. First there is a diversity-focussed approach, typified by the BIODEPTH experiment (Spehn *et al.* 2005), which showed that productivity in recently established grasslands was higher in more diverse plant mixtures than less diverse ones. This generated a significant volume of comment, much of which concentrated on the sampling effect (i.e. the higher underlying likelihood that the most diverse systems will contain the most productive species), and on reconciling this pattern with the well-established hump-backed relationship between species richness and productivity (Grime 1973).

An alternative approach focussed on plant traits and around the 'mass ratio hypothesis' of Grime (1998), which proposes that a species contributes to a function because of: (a) its contribution to biomass and; (b) the specific trait value related to that function. This boils down to the community weighted mean of a trait (CWM; the average value for a trait in a community weighted by the relative abundance of species) being a predictor of an ecosystem function. For example CWM leaf dry matter content is a predictor of leaf litter mass loss (Fortunel *et al.* 2009). This approach assumes that the traits of species control processes, and that adding additional diversity changes the trait mean or range of trait attributes, and does not affect function directly. This trait-based approach has recently been developed by de Bello *et al.* (2010) through the concept of trait-service clusters, i.e. the dependency of function and service on the traits of groups of organisms (clusters) in multiple trophic levels.

Interestingly, the two schools of thought outlined above have developed independently, and hence many experiments or reviews are set up to look uncritically at diversity effects on ecosystem function without looking at traits, whilst there has been an assumption – based on its apparent simplicity and elegance - that the 'mass ratio hypothesis' has to work. Most studies also stop at function: the links to services are not concrete, despite claims within some papers that it is services that are being considered (although this might stem in part from conflation of ecological processes and supporting services, as discussed in Section 1.4).

Studies that try to combine or reconcile the two paradigms are, however, becoming more common. For example:

- Díaz *et al.* (2007) explored the development of models that combine data on abiotic and trait characteristics of a system to explain variation of ecosystem service delivery in terms of land use and vegetation change. They showed that, for case studies based around alpine grassland systems, the primary driver of ecosystem function was either abiotic factors (which included the nitrogen nutrition index and above-ground biomass) or the CWM of traits. Variance in trait value was not important in controlling ecosystem function in these systems, and they conclude that "knowledge of FD [functional diversity] does not always decrease uncertainty with respect to EP [ecosystem processes] and that abiotic factors [e.g. above-ground biomass] can sometimes be important for practical purposes". The paper did not expressly test the impact of species diversity but did consider the influence of particular species or species groups and the functional diversity of species that might influence ecosystem function.

- Haines-Young & Potschin (2009) reviewed the literature and concluded that high diversity (species richness) tends to support resilience and sustained delivery of ecosystem services, but that thresholds are unpredictable. Thresholds occur when we shift from a state where a given decline in the status of biodiversity or biotic processes has a relatively limited impact on biodiversity, to one where we observe sudden declines in service delivery. Only more recently have resilience and threshold studies made explicit links to ecosystem service delivery; thresholds may help to specify limits to biodiversity degradation, but these may be only part of the solution, not least because biodiversity-ecosystem service relationships (and associated thresholds) may be context-dependent.
- The meta-analysis of Isbell *et al.* (2011) concluded that experimental systems may underestimate the potential functional importance of individual species, particularly when results are expanded across a broader suit of environments in time and space. Taking data from 17 grassland biodiversity experiments, they found that 84% of the 147 grassland species studied promoted ecosystem function at least once. The role of different species in promoting ecosystem functioning only become clear once multiple years, locations, functions and environmental change scenarios were taken into account. This analysis therefore cautions against assuming redundancy in function based on a limited range of contexts and/or timescales.

Some studies are now moving on to look at ecosystem service mapping. One example is the work by Lavorel *et al.* (2011) that tries to map ecosystem sservices through plant traits rather than through land use. This perhaps provides a more informative approach to understanding what underlies spatial variation in ecosystem service delivery.

However, although the work of Lavorel *et al.* scales up to the catchment level, most of the studies detailed above have been focussed on plot-level function. It is important to remember that the effects of biodiversity on ecosystem processes and service delivery at higher spatial scales may be more difficult to predict, and that patterns and relationships may be scale-dependent. For example Dakos *et al.* (2011) examined community properties which may act as indicators of thresholds between system states, including ecosystem collapse. They demonstrated that indicators for thresholds might not be universally applicable, but may depend on the unique characteristics of a given system.

A number of obvious questions and guiding principles for future work emerge from exploring this literature:

- We always have to assess the use of the term 'biodiversity'. Is it the 'diversity' of species present, is it a clumsy way of saying the 'biology' of a system, or is it just a proxy for the sampling effect, as functional richness is correlated to species richness?
- There is commonly a lack of clear identification in empirical studies of the precise function and/or ecosystem service being addressed.
- There are commonly trade-offs between services at a range of scales. Small (temporal or spatial)-scale studies may be unable to detect many such trade-offs. There is a need for larger-scale studies of the biodiversity-ecosystem service/function relationship explicitly considering scale as a critical component.

- Community level properties might be good indicators for system functions such as resilience, but what would the precise property be and can it act as a target for monitoring?

What is clear from the wide breadth of the literature, and the many different identified relationships between ecosystem function and biodiversity/plant traits/biophysical processes, is that the relationship is highly complex and context-dependent. The shape of the relationship could depend on the component of biodiversity examined (e.g. genetic-, species-, or habitat-level biodiversity), and the geographical location in which it is examined (e.g. semi-arid rangelands, upland heather moorland, or polar desert). It is critical, therefore, when trying to review the link between biodiversity and biophysical processes and ecosystem service delivery that we narrow down the context in which this link is being considered. We have tried to do this through the methodology adopted by this review.

1.4 Methodology for the Review

Given the wide breadth of information available, it was necessary to limit the scope of our review to provide more focussed outputs relevant to the Scottish context, rather than a more general discussion of the broad literature (as provided above). Other components of the Ecosystem Services Theme's synthesis and review phase, and other activities within the Ecosystem Services Theme, are also tasked with focussing their efforts on "key" ecosystem services, and so have also adopted this prioritisation approach. The prioritisation approach took two steps: selecting an ecosystem service categorisation, and then prioritising ecosystem services.

1.4.1 Ecosystem Service Categorisation

We chose to use the system for categorising ecosystem services that was developed by the UK NEA (Figure 1.1), not least because these are the ones with which many of the review's contributors and also most external UK stakeholders are already most familiar. The rationale for this categorisation of services is set out in full in Ch. 2 of the UK NEA. Use of this system restricts discussion about what constitutes a service, and enables us to relate our review directly to the very substantial body of information already collected by the UK NEA, as well as to other studies structured around the four service categories used within the MEA.

Figure 1.1. Diagram summarising the classification of ecosystem services as used in the UK NEA (redrawn from Table 2.2, UK NEA Ch. 2). Classification is according to both the ecosystem service type and whether or not they are final ecosystem services or intermediate services and/or processes. An example of the 'good' for each final ecosystem services is shown in italics.

<table>
<tr><th colspan="2">Ecosystem processes/intermediate services</th><th colspan="2">Final ecosystem services (examples of goods)</th></tr>
<tr><td>Supporting services</td><td>• Primary production
• Soil formation
• Nutrient cycling
• Water cycling</td><td>Provisioning services</td><td>• Crops, livestock, fish (food)
• Trees, standing vegetation, peat (fibre, carbon sequestration)
• Water supply (domestic and industrial water)
• Wild species diversity (bioprospecting, medicinal plants)</td></tr>
<tr><td colspan="2" rowspan="2">• Decomposition
• Weathering
• Climate regulation
• Pollination
• Disease and pest regulation
• Ecological interactions
• Evolutionary processes
• Wild species diversity</td><td>Cultural services</td><td>• Wild species diversity (recreation)
• Environmental settings (recreation, tourism, spiritual/religion)</td></tr>
<tr><td>Regulating services</td><td>• Climate regulation (equitable climate)
• Pollination
• Detoxification and purification in soils, air and water (pollution control)
• Noise regulation (noise control)
• Disease and pest regulation (disease and pest control)</td></tr>
</table>

In brief, within this conceptual framework ‘Final ecosystem services’ contribute directly to the ‘goods’ that are valued by people (and from which they derive a benefit): this covers regulating, provisioning and cultural services. ‘Intermediate ecosystem services’ and ‘ecosystem processes’ underpin the final ecosystem services but are not directly linked to ‘good(s)’ and “are less often the focus for management” (UK NEA Ch. 2): supporting services fall within this group. Supporting services do not themselves directly deliver benefits to people (Balmford *et al.* 2008) - benefits being “the end products of… ecosystem processes, which directly affect human wellbeing, and which can ultimately be evaluated economically (e.g. clean drinking water)”. Supporting services are excluded from final estimates of ecosystem service value. Such a categorisation strikes a balance between maintaining the MEA terminology whilst avoiding the risk of double-counting in ecosystem service valuation.

1.4.2 Focussing the Review using Broad Policy Goals

A variety of ‘goods’ may be delivered by a particular service. In turn the delivery of specific services might have highly context-dependent relationships with biodiversity and biotic and biophysical processes. Consequently, when considering how biodiversity and biotic and biophysical processes underpin the delivery of ‘goods’ or services, it is essential to define the context in which those ‘goods’ and services are being delivered. One form of context is policy. The success of certain policies is dependent on the delivery of particular ‘goods’ by particular services.

We considered the biodiversity/biophysical process–ecosystem service linkage within the context of a number of ‘broad policy goals’. This helped us to narrow the review’s focus and make our review outputs more directly relevant to current policy. These broad policy goals were developed from the *Scottish Land Use Strategy* (*Getting the best from our land: A land use strategy for Scotland*; Scottish Government, 2011), hereafter referred to as the LUS. The LUS sets out “a long term Vision towards 2050 with three clear Objectives relating to economic prosperity, environmental quality and communities.” In order to achieve this vision, and the associated objectives “the Strategy identifies key Principles for Sustainable Land Use which reflect Government policies on the priorities which should influence land use choices”.

From the LUS document, the Ecosystem Services Theme team formulated a set of five broad policy goals which it felt ran throughout the LUS. These are:

- Low Carbon Economy
- Sustaining Food Production
- Halting Biodiversity Loss
- Sustainable Water Management
- Enhancing Recreation Activities

1.4.2 The Prioritisation of Ecosystem Services

The prioritisation of ecosystem services important for delivering those policy goals was undertaken by a range of stakeholders at the second meeting of the Ecosystem Services Theme’s Ecosystem Approach Working Group (EAWG), held on 17th November 2011 at The James Hutton Institute, Invergowrie. Full details of the prioritisation process, and feedback on the process, are given in the EAWG report from the meeting (Eastwood *et al.* 2012[1]). In brief, a matrix of services against broad policy goals was presented to the participants. Participants were asked to each choose the five most important ecosystem services (considering both positive and negative policy implications) for

[1] http://www.hutton.ac.uk/research/themes/safeguarding-natural-capital/ecosystem-approach-working-group

delivering each of the five broad policy goals. The activity was initially focussed at the national level, and then moved on to focus on the Scottish uplands and arable lowlands.

The ecosystem services prioritised for each broad policy goal, at both the national scale and for upland and lowland systems, are set out in more detail at the beginning of each broad policy goal chapter. For illustrative purposes, Table 1.1 shows the top four prioritised ecosystem services under each broad policy goals at a national scale.

Table 1.1 *Top four prioritised ecosystem services under each broad policy goal, and the most highly ranked services overall (Total) at a national scale. Data taken from Eastwood et al. (2012).*

Low Carbon Economy	**Sustaining Food Production**	**Halting Biodiversity Loss**	**Sustainable Water Management**	**Enhancing Recreation Activities**
Trees	Crops	Wild species diversity (e.g. recreation)	Water cycling	Wild species diversity (e.g. recreation)
Peat	Livestock	Wild Species diversity (e.g. bioprospecting, medicinal plants)	Water supply	Environmental setting
Soil formation	Soil formation	Disease and pest regulation	Water detoxification & purification	Trees
Crops	Pollination	Crops	Hazard regulation	Fish

As can be seen from Table 1.1, prioritised ecosystem services differ between broad policy goals. For example crops are given top priority under Sustaining Food Production, whereas wild species diversity as a cultural service (for example in recreation) was most highly prioritised for both the Halting Biodiversity Loss and Enhancing Recreation Activities broad policy goals.

Focussing down the review does not make it parochial: by breaking down the work into simpler "bite-size" elements we start to see some interesting contrasts emerging between our understanding of different systems, and between different policy goals.

1.4.3 The Biodiversity and Biophysical Underpinning (BaBU) review process

The review process was structured around the broad policy goals and associated prioritised services, and these form the basis of the chapters which follow.

We did not pursue a chapter on Enhancing Recreation Activities. Although ostensibly about recreation, we decided that this broad policy goal was in fact trying to capture elements of the LUS that relate very strongly to cultural services, and health and wellbeing. Cultural services are a topic where there is currently substantial debate: the typology of cultural services is rapidly developing and it is hard at present to both define and measure many of them. Given these difficulties we did not think that we were in a position to then try to link these concepts to their biotic and biophysical underpinning. These concepts are instead being clarified by a specific piece of cultural services research being undertaken within Work Package 1.1 of the Scottish Government's Strategic Research Programme (specifically, Research Plan 5.3, Cultural Services of Woodlands).

Each broad policy goal chapter begins with a brief summary, followed by a description of the services prioritised for that broad policy goal. The chapters then consider how these prioritised services are supported by biodiversity and biotic and biophysical processes at a national scale, and then for upland and lowland systems. In this assessment, we have tried to consider the chain of linkages from biodiversity/biophysical processes all the way to the 'good'/benefit delivered to society, rather than stopping at either the ecological function or regulation of the service. In most cases we organised information under headings that reflect the prioritised services. However, it should be noted that we deviate from this general structure with respect to Ch. 5: Sustainable Water Management, for reasons that are discussed fully within the chapter.

In addition each chapter briefly assesses the potential for interactions between prioritised ecosystem services, and summarises key identified knowledge gaps. Interactions and knowledge gaps are then returned to in the final chapter (Ch.6), which summarises information from the broad policy goal chapters and puts this information within the broader context of research challenges and approaches that might be taken to address them.

Throughout we have applied a loose definition of upland and lowland systems: the distinction is really one of productivity, contrasting productive (e.g. arable, or intensively-stocked) lowland systems typical of the east coast and south west of Scotland, with the less productive, sometimes extensively grazed upland and mountain systems typical of northern Scotland and higher altitudes in the south. The latter, although considered "upland", have lower altitudinal limits further north and west and can reach sea-level, for example in the north-west Highlands (Fraser Darling & Morton Boyd 1964).

1.4.4 Comments on the prioritisation process

It is important to acknowledge that levels of satisfaction with the EAWG2 ecosystem service prioritisation process varied between participants. Below we have included in italics sections from the EAWG report (Eastwood *et al*. 2012.) detailing where participants disagreed with the prioritisation process. In each case we discuss how these concerns relate to the approach adopted by this review.

> *Eleven of the 19 participants that voted did not agree or only agreed in part to the prioritisation table. Eight of the participants can live with it, can support it or are very supportive of it. One of the key reasons for not agreeing or only part agreeing was that the most important ecosystem services were very dependent on the policy goal and system. The participants felt that one couldn't just tally across to get a list for the whole of Scotland. For example, water supply, hazard regulation and water detoxification are considered to be very important for sustainable water management but are in the bottom half of the prioritisation table for all the policy groups. This highlights that the relevance of ESS[*1] needs to be considered within the context of different policy or other goals.*
>
> [*1] The EAWG report uses ESS rather than ES as the short-hand for ecosystem services

This review considers each of the prioritised ecosystem services *within* their specific policy context, and contrasts the relationships in upland and lowland systems. In this way it actually explores in more depth this context-specificity.

> *A participant queried why we constrained ourselves to the Land Use Strategy (LUS), because if we chose a set of different policy goals e.g. a health policy, we would get a very different prioritisation.*

Other policy areas would certainly have given different prioritisations (as indicated by the differences in prioritisation rankings across the different broad policy goals). However, the LUS covers a wide range of separate policy areas that are highly relevant to land use and land use decision making. The policy areas encompassed within the LUS are likely to be key foci for the roll-out of an Ecosystem Approach and the ecosystem service concept.

> *In addition, the majority of the provisioning and cultural services - which tended to come out as most important in the prioritisation table - were closely linked to, underpinned, or dependent on the supporting and regulating services lower down the table.*

The low scoring of supporting services is very clear. This may reflect the focus of activities such as UK NEA service valuation on final ecosystem services (to avoid double counting, as mentioned above). Where considered important, some key supporting services have been discussed in some of the broad policy goal chapters in this review. In addition, consideration of the cascade running from biodiversity or biotic/biophysical processes to final ecosystem services inevitably has to incorporate consideration of supporting services and processes (as shown in Fig. 1.1). So these services are not overlooked.

> *A few participants felt that the cultural services presented were too narrow in scope and should be expanded to be more representative of their importance.*

As noted above, cultural services are an area of considerable complexity. Although, because of this, we have not included a chapter on Enhancing Recreation Activities in the review, some cultural services are dealt with in the chapters looking at the other broad policy goals. We have not, however, altered the terminology for considering cultural services. As mentioned, this is an area of current research activity; we have utilised the UK NEA terminology throughout rather than trying to encompass new typologies emerging from this fast moving field.

> *A number of participants commented that we should be consulting stakeholders other than research scientists* [those involved in research directly or policy development]*, particularly those working on the ground such as land managers, estate owners and farmers. As one participant commented, this is particularly relevant if applying the Ecosystem Approach (CBD, 2011) which is meant to take on board society's views with regards to natural resource management. This was only briefly discussed as wider involvement of stakeholders is envisaged within subsequent stages of the research programme.*

As noted above, wider consultation of a full range of stakeholders will take place throughout the 5 year lifetime of the Ecosystem Service Theme. Importantly, we do not propose that the prioritisation used in this review is the final word on defining the key ecosystem services for Scotland, but we do believe that the prioritised services are important and valuable foci for this review. Notably, "*Those that were supportive of the prioritisation table considered food production and security to be the biggest challenge facing us in the next 20 years*", and this is reflected in the prioritised services. We plan to return to this review toward the end of the work of the Ecosystem Services Theme, and revise it in the light of new knowledge acquired.

> *Another approach to prioritisation was suggested by several EAWG members that were unable to come to the workshop. They suggested that prioritisation should be based on the benefits to humans of each ESS* [ecosystem service] *and not what we (EAWG members) think is important. The ESS ranked most highly may simply be the ones that we just know the most about, whereas maybe we should focus our attention on the things we know less about. A counterpoint is that the alternative approach might simply rank highly those services that are easy to value, and not necessarily those that are important. This needs to take into consideration that there is incomplete understanding of the links between value and 'importance'.*

Either approach to prioritisation has plus and minus sides. The final prioritisation based on "importance", which may encompass more than readily-measured benefits, appears to give useful results.

> *A number of participants asked why we were using current policies, such as those in the LUS, to help us select ESS on which to focus. They argued that it should be the other way round: research should be used to understand the importance and interactions of ESS for human wellbeing, which then informs policy. In response to this argument it was recognised that the use of research to inform policy and vice versa was an iterative process, with new policies being built upon new research findings. The success of such an iterative process is determined by the engagement and dialogue between policy makers and research scientists, which inevitably must start at some point within the cycle (in this case taking the policy rather than the science as the basis for discussion). The EAWG can provide an important forum for this dialogue to occur.*

In the context of this review, the policy framework used in EAWG 2 provides us with a useful set of contrasting contexts within which to examine the biodiversity/biophysical process–ecosystem service relationship.

1.5 Working Terminology for the BaBU Synthesis and Review

From the initial stages of the review it became clear that for cohesion and internal consistency all chapter teams should try to work to consistent definitions of key concepts. These definitions are set out below. Although we hope they capture the key elements of these concepts as they are currently being applied, even if our definitions differ from those used elsewhere, being explicit here at least enables the reader to map between studies. Some of these concepts have already been discussed in detail above. They are presented here for ease of reference. Excellent glossaries are presented within the UK NEA, and by Mace *et al.* (2012), and we have used these as the basis for several definitions.

Adaptation (Evolutionary)

Adaptation is widely used to refer to human actions – individual, or social, planned or unplanned – which mitigate vulnerability to global environmental change. However, confusion can and does occur in discussing adaptation, especially at the interface of policy and biodiversity. This is because biologists use adaptation to refer to the process of evolutionary change in a species' phenotype, which acts to maintain ecological fitness by means of natural selection. This confusion may be compounded because the process of evolutionary adaptation to environmental drivers (e.g. climate change), has implications for biodiversity conservation, and informs the alternative definition of strategic adaptation defined by human actions. Here, when we are referring to the process of evolutionary adaptation by natural selection we will state this explicitly.

Biodiversity and Biotic Processes

Article 2 of the CBD provides a widely-used definition of biodiversity:

> '*Biological diversity' means the variability among living organisms from all sources including,* inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems*.

Biodiversity within the CBD is not therefore an inventory of biological entities (a count of different genes, species, or ecosystems) but represents the variability within and among these things.

Biodiversity can appear at all levels across the ecosystem service hierarchy: "as a regulator of underpinning ecosystem processes, as a final ecosystem service and as a 'good' that is subject to valuation, whether economic or otherwise" (Mace *et al.* 2012). It can also be used to refer to "nature" or natural systems. In this review:

- 'Biodiversity' is reserved strictly for discussing the diversity of biota at whatever level in the genes-to-habitats hierarchy, and we specify the level of diversity being discussed.
- When discussing simply nature, biota, or biotic/natural processes, rather than something that is specifically about *diversity*, we use these other phrases instead.

Biophysical processes

Biophysics is the science of the application of the laws of physics to biological phenomena. However, when applied in an ecological context the term has a slightly different meaning: the biophysical environment is the combined biotic and abiotic environment of an organism.

In this review the biophysical environment is then taken as encompassing the elements of the environment beyond biodiversity and biotic processes. Hence we take biophysical processes as including other potential environmental regulators of ecosystem service delivery such as non-organismal soil processes. In addition we explicitly distinguish between biophysical processes and land form. Land form is an element of the natural environment that can strongly influence service delivery, but which is itself not a process.

The Ecosystem Approach

The CBD defines 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.*

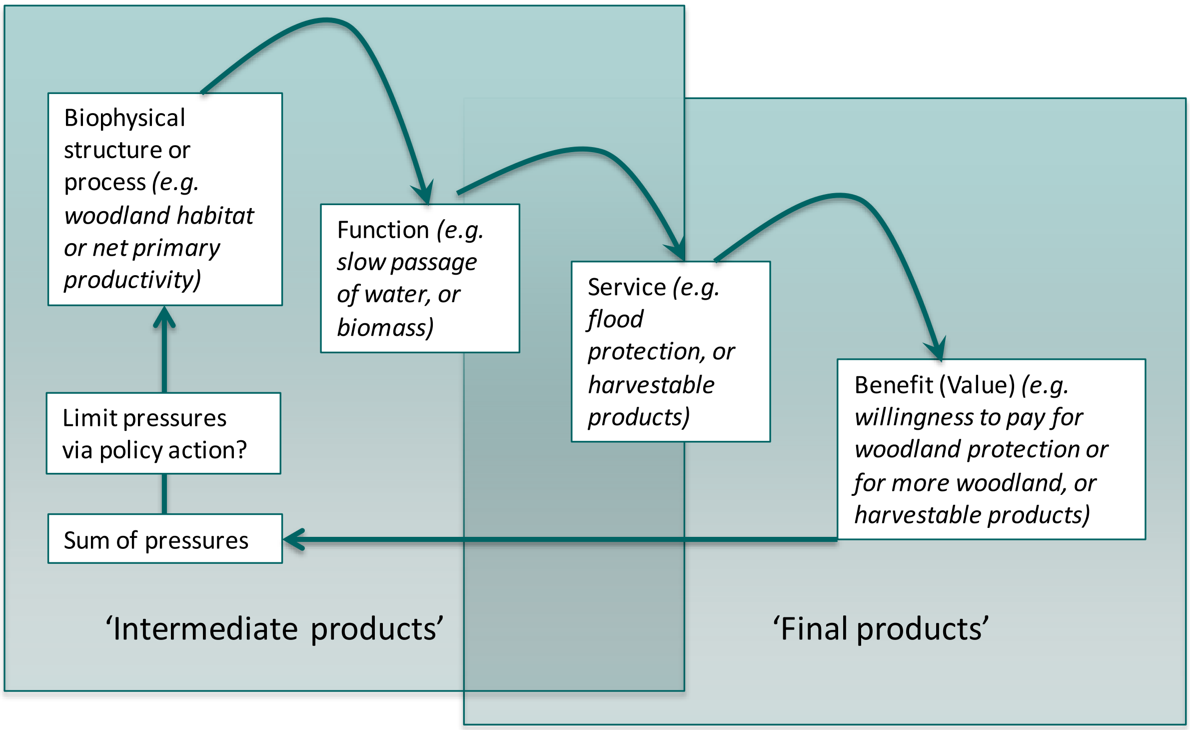
This is underpinned by the 12 CBD Principles (Appendix 1).

Ecosystem services

Is an "activity or function of an ecosystem that provides benefit (or occasionally disbenefit) to humans". Final ecosystem services directly underpin or give rise to a 'good'. (Mace *et al.* 2012).

The link between biodiversity, ecosystem function, and human well being is sometimes conceptualised as an ecosystem service cascade such as that described by Haines-Young &Potschin (2010), which takes the structure shown in Fig. 1.2.

Fig 1.2. The ecosystem service cascade. Redrawn from Haines-Young &Potschin (2010).



Functional Groups/Types and Functional Diversity

In ecology, functional groups have been defined as "sets of species showing either similar *responses to* the environment or similar *effects on* major ecosystem processes"; the terms functional groups and functional types are often used synonymously (Hooper *et al.* 2002, after Gitay & Noble 1997).

Functional diversity is "the range and value of organismal traits that influence ecosystem properties" (Hooper *et al.* 2002, after Tilman 2001).

'Goods'

'goods' are "the objects from ecosystems that people value through experience, use or consumption" (Mace *et al.* 2012).

Natural Capital

In economics, capital is a 'good' produced by labour, which yields utility through its performance in the production process, leading to downstream consumed services and 'goods' over a long-period.

Borrowing from, and modifying this economic concept, natural capital is defined as the stock of functional components (biodiversity elements) occurring within ecosystems, and which yield a provision of services and flow of 'goods'.

Sustainability and sustainable development

This is a much-discussed term. However, a recent POST Note (POST, 2012) sets out in detail the various uses and interpretations of the terms sustainability and sustainable development. It has been used here as the basis for deriving the following definitions:

- Sustainability is the long term maintenance and enhancement of human well-being within finite planetary resources. It requires integrating environmental resilience with human well-being, incorporating a long term perspective.
- Sustainable development is often used interchangeably with sustainability. In 1987, the World Commission on Environment and Development defined SD as the "development that meets the needs of the present without compromising the ability of future generations to meet their own needs". SD is the framework, process, or group of processes for integrating environmental, social and economic factors within all policy decisions, to develop the most sustainable policy option.

United Kingdom National Ecosystem Assessment: Ecosystem Service Categorisation

The categorisation of ecosystem services within the UK NEA follows the MEA in aiming to specify the processes and/or properties of an ecosystem from which 'goods' (use and non-use, material and non-material outputs from ecosystems that have value for people) are derived.

For example, biomass is a property of an ecosystem, though the biomass of trees may represent the specific ecosystem service, and timber the 'good'. Pollination is an ecosystem process, with recurrent seed-setting the ecosystem service, and a crop of apples the 'good'. These properties and processes are sometimes referred to as 'intermediate ecosystem services'.

The ultimate aim is to allow for an economic assessment, in which the monetary or non-monetary value of a 'good' can be partitioned into a component which is freely derived from the ecosystem, compared to managed interventions in deriving the same 'good'(s). Ecosystem services are of four broad types: supporting services, provisioning services, cultural services and regulating services.

Valuation and Values (from the UK NEA)

Valuation: The process of expressing a value for a particular 'good' or service in a certain context (e.g. of decision-making) usually in terms of something that can be counted, often money, but also through methods and measures from other disciplines (sociology, ecology, and so on). See also *Value.*

Value: The contribution of an action or object to user specified goals, objectives, or conditions. (Compare *Valuation*).

Values - monetary: Measurements of the value associated with ecosystem services that are expressed in monetary units. This measurement can be obtained from observation of the prices or costs of transactions, i.e. *market valuation*. Alternatively, *non-market valuation* uses either proxy markets to infer value from observed behaviour (*revealed preferences,* e.g. *travel cost, hedonic price models*), or hypothetical markets (*stated preferences,* e.g. *contingent valuation, choice experiments*).

Values - non-monetary: Measurements or motivations for value that are not expressed in monetary units. These might include proxy measures such as travel time (which can be converted to monetary values). The UK NEA makes a specific distinction between economic (monetary) values and health and shared social values. ***Health values*** consider the (human) *physical, emotional and social* well-being associated with the state of a particular ecosystem (and its services) or the change in that well-being brought about by ecosystem change. ***Shared social values*** include wider 'values' such as ethical, cultural and aesthetic concerns that may be held by people as *citizens* rather than *individuals*. These *values* can be obtained through observation (health values) or through multicriteria and deliberative approaches (shared social values).

References

Balmford, A., Rodrigues, A., Walpole, M. J., ten Brink, P., Kettunen, M., Braat, L. & de Groot, R. (2008) *Review of the Economics of Biodiversity Loss: Scoping the Science.* European Commission, Brussels.

Balvanera, P., Pfisterer, A.B., Buchmann, N., He, J-S., Nakashizuka, T., Raffaelli, D. & Schmid, B. (2006) Quantifying the evidence for biodiversity effects on ecosystem functioning and services. *Ecology Letters*, **9**, 1146-1156.

Constanza, R., Fisher, B., Mulder, K., Liu S. & Cristopher T. (2007) Biodiversity and ecosystem services: A multi-scale empirical study of the relationship between species richness and net primary production. *Ecological Economics*, **63**, 478-491.

Dakos, V., Kéfi, S., Rietkerk, M., Van Nes, E. H. & Scheffer, M. (2011) Slowing down in spatially patterned ecosystems at the brink of collapse. *The American Naturalist*, **177**, E153-E166.

de Bello, F., Lavorel, S. Díaz, S. Harrington, R., Cornelissen, J.H.C., Bardgett, R.D., Berg, M.P., Cipriotti, P. Feld, C.K., Hering, D. da Silva, P.M., Potts, S.G. Sandin, L. Sousa, J.P., Storkey, J., Wardle, D.A. & Harrison, P.A. (2010) Towards an assessment of multiple ecosystem processes and services via functional traits. *Biodiversity and Conservation*, **19**, 2873-2893.

Díaz, S., Lavorel, S., de Bello, F., Quétier, F., Grigulis, K. & Robson, T.M. (2007) Incorporating plant functional diversity effects in ecosystem service assessments. *Proceedings of the National Academy of Sciences*, **104**, 20684-20689.

Eastwood, A., Black, H., Irvine, J., Mcvittie, A., Brooker, R., Waylen, K., Coull, M., Hastings, E. & Banks, E. (2012) EAWG2 Discussion Paper: Prioritising Ecosystem Services and Appraising Indicators. James Hutton Institute, Aberdeen and Dundee.

http://www.hutton.ac.uk/research/themes/safeguarding-natural-capital/ecosystem-approach-working-group

EEA (2005) *Environmental policy integration in Europe. State of play and an evaluation framework.* EEA technical report no. 2/2005. European Environment Agency, Copenhagen.

Fortunel, C. ,Garnier, E., Joffre, R., Kazakou, E., Quested, H., Grigulis, K. *et al.* (2009) Leaf traits capture the effects of land use changes and climate on litter decomposability of grasslands across Europe. *Ecology*, **90**, 598–611.

Fraser Darling, F. & Morton Boyd, J. (1964) *The Highlands and Islands*. Collins, London.

Gitay, H. & Noble, I.R. (1997) What are functional types and how should we seek them? *Plant Functional Types* (eds T.M. Smith, H.H. Shugart, & F.I. Woodward), pp. 3-19. Cambridge University Press, Cambridge.

Grime, J.P. (1973) Competitive exclusion in herbaceous vegetation. *Nature*, **242**, 344–347.

Grime, J.P. (1998) Benefits of plant diversity to ecosystems: immediate, filter and founder effects. *Journal of Ecology*, **86**, 902–910.

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.

Haines-Young, R. & Potschin, M. (2009) *Environmental Limits, Ecosystem Resilience and Supporting Services*. LWEC Pilot Review. Contract Number: R8/H12/106.

Haines-Young, R. & Potschin, M. (2010) The links between biodiversity, ecosystem services and human well-being. *Ecosystem Ecology: a new synthesis* (eds D. Raffaelli & C. Frid), pp.110-139. Cambridge University Press, Cambridge.

Hooper, D.U., Solan, M., Symstad, A., Diaz, S., gessner, M.O., Buchmann, N. *et al.* (2002) Species diversity, functional diversity, and ecosystem functioning. *Biodiversity and Ecosystem Functioning: Synthesis and perspectives* (eds M. Loreau, S. Naeem & P. Inchausti), pp. 195-208. Oxford University Press, Oxford.

House of Commons Environmental Audit (2007) *The UN Millennium Ecosystem Assessment*. First Report of Session 2006-7; HC77, 58pp.

Howard, B.M., Hails, R.S., Watt, A., Potschin, M. & Haines-Young, R. (2011) *Considerations in environmental science and management for the design of natural asset checks in public policy appraisal.* Paper presented at a workshop hosted by Defra, 11th May 2011. Defra Project Code NE0122.

Isbell, F., Calcagno, V., Hector, A., Connolly, J., Harpole, W.S., Reich, P.B. *et al.* (2011) High plant diversity is needed to maintain ecosystem services. *Nature*, **477**, 199-202.

Lavorel, S., Grigulis, K., Lamarque, P., Colace, M.-P., Garden, D., Girel, J., Pellet, G. & Douzet, R. (2011) Using plant functional traits to understand the landscape distribution of multiple ecosystem services. *Journal of Ecology*, **99**, 135-147.

Mace, G.M., Norris, K. & Fitter, A.H. (2012) Biodiversity and ecosystem services: a multilayered relationship. *Trends in Ecology and Evolution*, **27**, 19-26.

Millennium Ecosystem Assessment (2005). *Ecosystems and Human Well-being: Synthesis*. Island Press, Washington, DC.

POST (2012) *POST Note 408: Seeking Sustainability*. Parliamentary Office of Science and Technology, London**.** www.parliament.uk/post

Scottish Executive (2004) *Scotland's Biodiversity: It's In Your Hands. A strategy for the conservation and enhancement of biodiversity in Scotland*. Scottish Executive, Edinburgh.

Scottish Government (2011) *Getting the best from our land: A land use strategy for Scotland.* The Scottish Government, Edinburgh.

Secretariat of the Convention on Biological Diversity (2000) *How the Convention on Biological Diversity promotes nature and human well-being.* Secretariat of the Convention on Biological Diversity with the support of the United Nations Environment Programme (UNEP) and the Government of the United Kingdom. SCBD, Montreal.

Secretariat of the Convention on Biological Diversity (2010) *Global Biodiversity Outlook 3*. SCBD, Montréal.

Spehn, E.M., Hector, A., Joshi, J., Scherer-Lorenzen, M., Schmid, B., Bazeley-White, E. *et al.* (2005) Ecosystem effects of biodiversity manipulation in European grasslands. *Ecological Monographs*, **75**, 37-63.

Tilman, D. (2001) Functional diversity. *Encyclopedia of Biodiversity, Vol. 3* (ed S.A. Levin), pp. 109-120. Academic Press, San Diego.

Chapter 2: Low Carbon Economy

2.1 Summary

A low carbon economy is described in the Scottish Land Use Strategy (LUS; The Scottish Government 2011) as "An economy in which less energy and resources are used" and "where energy increasingly comes from sources that produce fewer carbon emissions". A low carbon economy promotes a reduction in the use of energy and resources, greater use of renewable energy, reduced carbon emissions and increased carbon storage. Trees, peat, soil formation and crops were considered by the Ecosystem Approach Working Group EAWG 2 workshop attendees as the most important ecosystem services with respect to a low carbon economy.

Within the UK NEA, trees, peat and crops are defined as provisioning and regulating services and soil formation is a supporting service. As provisioning services trees, peat and crops provide a range of 'goods' including food, timber and energy. In relation to a low carbon economy it is the regulating service of carbon sequestration provided by trees, peat, crops and soil formation that is important. The carbon sequestration provided by trees, crops, peat and soil formation contribute to the final ecosystem service of Climate Regulation and the 'good' is a more equitable climate. In relation to crops the 'goods' of fibre, bioethanol and biodiesel, along with more general food and fibre 'goods' produced by reduced input farming, are the 'goods' of greatest relevance to the broad policy goal of a low carbon economy.

With respect to trees:

- Biophysical conditions will influence which, if any, tree species will grow and the rate of growth and hence carbon storage. The limitations imposed by biophysical conditions may, to some extent, be overridden by management practices such as fertilisation and drainage. However such practices are often in conflict with methods to promote a low carbon economy.
- Biotic processes may enhance the ecosystem service of trees through increased growth (e.g. via mycorrhizal associations), pollination, dispersal of seeds/fruits and pest regulation. Biotic processes may negatively influence the ecosystem service of trees through diseases and pests and browsing by herbivores which reduces tree growth/survival. An increase in the ecosystem service of trees (i.e. increased tree cover) will result in more stored carbon in the vegetation, but this must be balanced againstthe possible loss of carbon in the soil resulting from changes in soil properties and soil function as woodland establishes.
- A combination of abiotic and biotic factors together with forestry management practices will influence the total amount of carbon stored by trees.

With respect to peat and soil formation:

- The extraction of peat as a 'good' for fuel and horticulture (totallingtotalling 265,000 m^3 in Scotland in 2008) inevitably leads to GHG emissions, and is in conflict with the 'good' of an equitable climate resulting from carbon sequestration delivered by peat.
- Peat formation is underpinned by the biotic processes of the peat forming plants- and associated litter-decomposing microbial communities. Since *Sphagnum*-dominated bryophyte communities are the main peat builders in most high-latitude peatlands, *Sphagnum* productivity and decomposition rates are important for evaluating changes in peat formation and carbon sequestration, but there is still considerable uncertainty about the influence of associated decomposer biodiversity on peat formation.
- The majority of soil processes, including many of those involved in soil formation, are mediated by the activity of the soil microbial community. Changes in plant community structure will affect microbial community structure and diversity. Increases in microbial diversity are generally associated with more diverse plant communities.

- The huge biological diversity found in soils may be largely redundant with respect to soil processes. However, 'inactive' diversity may be central to system or functional resilience, and discounting this diversity would be unwise without a full understanding of the soil environment and its processes.

With respect to crops:

- Crops play a major role in delivery of a low carbon economy because land used for crops might be diverted into alternative non-crop land uses that support renewable energy rather than food production, because the type of crops grown on cropped land might change to accommodate increased demand for particular energy crops, or because changes in crop management (e.g. reduced input farming) might lead to changes in carbon emissions.
- Shifts in land management have the potential to alter biodiversity at field and landscape scale through changes in nutrient and chemical inputs and in farm, soil, weed and pest management.

Overall:

- The importance of trees, peat and crops as a regulating service (carbon sequestration) and provisioning service (fibre, food) and soil formation as a supporting service will vary between upland and lowland habitats. Upland habitats generally provide more carbon storage (trees and peat) than lowland habitats, while lowland habitats generally provide more food and fibre. However these broad generalizations hide much local variation, for example the type and quality of the crop will vary with climate, altitude and aspect, which influence the resulting goods.
- Within the ecosystem services prioritised for a low carbon economy, the most common conflicts occur around land use and land management. Decisions have to be made over which crops (food or biofuels), management practices, or habitats (woodland or other habitats) to have on any given piece of land, with an acknowledgement that such decisions will influence the success of policies to implement a low carbon economy. Ecosystem service mapping is invaluable in this respect, allowing spatially explicit strategic visioning, explicit consideration of trade-offs, and decision-making.
- There are many scientific gaps in our understanding about how biodiversity, biotic and biophysical processes underpin the delivery of ecosystem services relevant to achieving a low carbon economy.

2.2 Definition of a low carbon economy

A low carbon economy is described in the Scottish Land Use Strategy (LUS; The Scottish Government 2011) as "An economy in which less energy and resources are used – domestically, commercially and across the public sector; where energy increasingly comes from sources that produce fewer carbon emissions, such as water, wind, wave and solar power; and where economic opportunities from efficiencies and saving carbon are realised." Carbon storage, although not included within the above definition, is also part of a low carbon economy and is discussed throughout the LUS.

A strategy for a low carbon economy was produced by the Scottish Government in 2010 (The Scottish Government 2010a) as part of the Government's response to its climate change commitments, and describes the benefits and opportunities of building a low carbon economy in Scotland. The strategy is composed of four documents. Within these four documents the following two sections are most relevant for biotic and biophysical processes: the "Resources" section in "*A Low Carbon Economic Strategy for Scotland*" which focuses on economic opportunities, and chapter 7 "*Rural Land use*" within "*The report on proposals and policies*" which sets out specific measures for reducing greenhouse gas emissions to meet Scotland's statutory targets. These sections outline the key biological processes that are CO_2 sources, and the land uses and practices across all sectors that contribute to carbon storage.

There are a number of other Scottish Government policies, developed in line with the low carbon economy goal, that help to take forward its delivery. Of relevance to biotic and biophysical processes are those policies on land-use, soils, agriculture and forestry: *The Land Use Strategy* (The Scottish Government 2011), *A Vision for Scottish Agriculture* (The Scottish Government 2010b), the *Scottish Soils Framework* (The Scottish Government 2009a), and *The Scottish Forest Strategy* (Scottish Executive 2006; The Scottish Government 2010c). The Land Use Strategy demonstrates the role of land use in a low carbon economy, addressing issues such as reducing greenhouse gas emissions from land, capitalising on renewable energy opportunities, carbon retention in peat soils, and balancing goals for food security and tree-planting. As part of the low-carbon economy, the Scottish Government has committed to increase Scotland's woodland cover from 17% to 25 % to realise a range of benefits including carbon sequestration (Scottish Executive 2006). To achieve this there are policies for woodland expansion (The Scottish Government 2009b) and targeted planting so as to maximise carbon capture and minimise losses from the soil (Forestry commission 2003; Forestry Commission Scotland 2010). In June 2012 the Woodland Expansion Advisory Group published their report on where the desired increase in woodland cover will occur (Woodland Expansion Advisory Group 2012). This report recommended 10,000 ha to be planted annually between 2012 and 2020. Programmes such as Farming for a Better Climate run by the Scottish Agricultural College (now SRUC), and a *Vision for Scottish Agriculture* (The Scottish Government 2010b), are both aimed at encouraging farmers to farm in a way that contributes to a low carbon economy through practices such as using energy and fuels efficiently, developing renewable energy, locking carbon into the soil and vegetation, optimising the application of fertiliser and manures, and optimising livestock management and storage of waste. The Scottish Soils framework identifies 13 outcomes with respect to soil, many of which will contribute to a low carbon economy. Thus a low carbon economy "philosophy" is woven into many of the recent Scottish Government's policies.

2.3 Prioritized ecosystem services

The ecosystem services prioritized by the Ecosystem Approach Working Group for a low carbon economy were trees, peat, soil formation and crops. Trees, peat and crops are considered as provisioning and regulating services in the UK NEA, and soil formation a supporting service. The carbon sequestration provided by trees, crops and peat is a regulating service contributing to the final ecosystem service of Climate regulation and the 'good' is a more equitable climate. As provisioning services trees, peat and crops provide a range of 'goods' including food, timber and energy. How these 'goods' are used (e.g. use of timber instead of more energy intensive materials such as concrete) or the decision not to use the 'goods' (e.g. peat as a fuel) are all part of a low carbon economy. In relation to crops the 'goods' of fibre, bioethanol and biodiesel, along with more general food and fibre 'goods' produced by reduced input farming, are the 'goods' of greatest relevance to the broad policy goal of a low carbon economy.

With respect to a low carbon economy, the regulating service of carbon sequestration provided by trees, peat and crops is crucialcrucial. In this sense, soil formation might also be considered a regulating service: its primary function with respect to a low carbon economy being carbon sequestration. The regulating service of carbon sequestration results in the 'good' of a more equitable climate and it is the value of this equitable climate that is assessed when carbon sequestration is valued.

The importance of trees, peat and crops as a regulating service (carbon sequestration) and provisioning service (fibre, food) and soil formation as a supporting service will vary across Scotland, particularly between upland and lowland habitats. Upland habitats generally provide more carbon storage from trees and peat than lowland habitats, while lowland habitats provide more food and fibre. However these broad generalizations hide much local variation.

2.3.1 Trees

Trees provide humans with a range of 'goods': timber (plus associated products made of timber such as paper), fuel in the form of logs or wood pellets, Christmas trees, edible non-timber forest products (predominantly the wild mushroom industry, but also venison, berries), fruit from orchards, and shelter for houses and livestock (UK NEA Ch. 2 and 15). In addition to their contribution as a provisioning final ecosystem service, the UK NEA recognises the roles of trees as a regulating service (air quality, sequestering carbon, UK NEA Ch. 14), a cultural service (aesthetic, UK NEA Ch. 8) and a supporting service (primary production). With respect to a low-carbon economy the service of sequestered carbon is of most relevance. The removal of carbon dioxide from the atmosphere is a regulating service which is carried out by all plants. However in this chapter we focus in particular on the regulating service provided by trees through carbon sequestration. It is noted that in the UK NEA that the good "carbon sequestration" provided by trees is variously defined as being a provisioning service (UK NEA Table 2.2) or a regulating service, (UK NEA Ch. 14.2). Here we describe carbon sequestration as a regulating service in order to be consistent with its use in the majority of the UK NEA.

Although sensitive to the assumed social values of carbon (and to assumptions of permanence), the NEA suggests a carbon storage value of £239 ha^{-1} yr^{-1} for woodlands, compared to a mean value for softwood production of £66 ha^{-1} yr^{-1} and £7-£25 ha^{-1} yr^{-1} for hardwood production. However carbon sequestration remains a largely non-market commodity, and there is little incentive at present for landowners to increase the delivery of this ecosystem service (or to maintain existing carbon storage). In addition, the use of wood fuels as a substitute for fossil fuels and the use of timber and wood products, instead of more energy intensive materials such as concrete and steel, also contribute positively to a low carbon economy (Scottish Government 2010a).

Trees occur in both the uplands and lowlands, and so tree-sequestered carbon is distributed across both environments, although is absent from more mountainous upland areas above the limit of tree growth. The capacity of uplands and lowlands to deliver carbon sequestration through trees will vary depending on tree species (how suited a species is to growing in a given area's climate) and underlying levels of productivity. There was an increase in commercial plantations in the uplands in the 1980s, but this has since virtually stopped as the negative effects on biodiversity and soil carbon in some soils have been realised: when trees are planted on soils with high carbon content they dry out the soil resulting in less carbon being stored. However, there remains a large area of commercial forestry in many upland areas. In June 2012 the Woodland Expansion Advisory Group published their report on where the desired increase in woodland cover will occur (Woodland Expansion Advisory Group 2012). This report identified 46% of Scotland's land as largely unavailable for woodland creation (it is unsuitable, already wooded or ruled out by policy considerations); a further 20% is significantly constrained (in particular because of conservation designations). The remaining one third of Scotland's land has the most potential for woodland creation, with this land occurring in both upland and lowlands, although it is predominantly farmland (open grazing land) that was identified (Woodland Expansion Advisory Group 2012).

2.3.2 Peat

In the UK NEA, peat is considered as a provisioning ecosystem service, providing a 'good' (peat) that is used for fuel and horticulture, (if undisturbed) a regulating service through carbon sequestration, and a cultural service through the delivery of a distinctive landscape with its associated land use traditions (peat cutting) and wildlife/botanical interest. Historically, extracted peat was used for domestic fuel and stable litter. Although some peat is still used for domestic fuel, most of the peat extracted today is used in the whisky and horticultural industries. In the whisky industry, peat is used to dry damp malt, thereby adding to the flavours of malt whiskies, although the NEA reports the intention within the industry to reduce its use of peat. The use of peat in the horticultural industry peaked in the 1990's at 96% of market share. However, as a result of government pressure the average peat content in growing media was reduced to 81% and 72% in professional and retail products, respectively, while peat in soil improvers accounted for just 2% of sales by volume by 2007. The trend in reduced use and market proportion of horticultural peat is reflected in total volume extracted and land area used. The total land area fell from 14,980 ha in 1994 to 10,690 ha in 2009. The total GB extracted peat volume was more than halved from 1999 to 2008 (1,616,000 m^3 to 760,000 m^3, respectively), whereas the Scottish output fell by one third (392,000 m^3 to 265,000 m^3; UK NEA Ch. 15).

The provisioning, regulating and cultural services of peatland ecosystems are discussed in detail in the UK NEA with respect to moorland (Ch. 5) and wetland habitats (Ch. 9). The use of peat for fuel and horticulture is intricately linked to the functioning and existence of the peat-forming habitats. The use or conservation of peat in the context of a low carbon economy inevitably has consequences for these habitats. Thus, although we can discuss the use or conservation of peat simply with respect to the implications for C cycling and storage, in reality there will inevitably be much wider consequences for these entire peat-forming ecosystems. Some of these will in turn feed-back on the goal of a low carbon economy, whilst others will impact on the delivery of other ecosystem services and broad policy goals. For example, a recent review of current UK policies of managing peatland ecosystem services by Whitfield *et al*. (2012), and a DEFRA commissioned report (Bonn *et al*. 2009), integrate peat-based provisioning services into the wider framework of the peatlands habitat ecosystem services. Peat formation takes hundreds to thousands of years: peat depth is estimated to increase at a rate of up to 0.8 mm/yr in actively growing bogs of good status and ideal environmental conditions (UK NEA Ch.19). It is therefore generally regarded as a non-renewable energy source. Accordingly, the consumption of peat for fuel and horticulture leads to net increased atmospheric GHG (greenhouse gas) concentrations. This is exacerbated by the concurrent draining of peatlands which is required for commercial peat extraction, or major land use change such as tree planting. Peatland draining inevitably results in destruction of a carbon sequestering ecosystem and in substantial losses of sequestered carbon by peat degradation and export of particular and dissolved organic carbon (POC and DOC), thereby resulting in substantially higher CO_2 emissions than projected from extracted peat volume alone. The further use of peat for fuel and horticulture therefore seems incompatible with the goal of a low carbon economy, and impacts directly and negatively on sequestered carbon, resulting in a degradation of this ecosystem service.

A phase-out for peat-based compost by 2020 has been announced, however, there is still considerable resistance within the horticultural industry to accept peat alternatives within a competitive unregulated international market (Response from the Horticultural Trades Association Press Office 2011; Alexander *et al.* 2008). Due to increasing fossil fuel prices, peat fuels again have become a competitive alternative for domestic, municipal or small scale industrial use. Especially in peat-rich countries where decentralized energy production is common practice, peat fuel has maintained a significant part of the energy sector. In Finland, about 6.6% of the total primary energy consumption was produced with peat during 2006-2007, with 20.7% in municipal combined heat and power plants (CHIP), requiring 69500 ha of peat land for extraction (Paappanen *et al.* 2010). The competitive low price of peat per MWh heat and energy production (10.7€ peat; 35.6€ & 29.7€ heavy fuel oil; 27.0€ & 24.9€ gas; heat and electricity, respectively, Paappanen *et al.* 2010) could result in further increases in domestic or municipal peat fuel use.

"The Scottish Government is committed to increase the amount of electricity generated from renewable sources to 50% by 2020 and 11% of heat demand to be met from renewable sources. Production of heat and electricity from renewable sources will also make an important contribution both at a domestic scale and through decentralised energy and heat supply systems including district heating and biomass heating plants for businesses, public buildings and community/housing schemes" (The Scottish Government 2010d). In this context, the conversion of drained peatland of low economic and agricultural value for the production of biofuel crops or wind farm developments (which may prevent peatland re-wetting if not designed well) instead of regeneration schemes might prove attractive as the price of fossil fuels increases (Stunell *et al.* 2010). The negative aspect of further soil carbon loss and GHG emissions might appear acceptable by the "softening" promise of renewable energy production.

2.3.3 Soil formation

*S*oil formation is defined in the UK NEA as "the formation and degradation of the UK soil resource" and can be seen primarily as a supporting service, underpinning the provisioning services of crops, trees, water supply and wild species diversity, and a regulating service (see below). As such, soil formation is at the heart of the primary production industries of agriculture and forestry.

As a provisioning service, Scotland's soils hold 40 billion cubic metres of water in the top metre when fully wet (more than all the fresh water lochs). These soils also deliver key regulating services since they filter, transform and break down the hundreds of different acidifying, potentially toxic elements and compounds that enter ecosystems from polluted rain (as discussed in more detail in Ch. 5 of this review).

In terms of the low carbon economy, it is perhaps the delivery of significant carbon sequestration by both peat and soil formation (a regulating service) that contributes most to human well-being through mitigating climate change. Scotland's peatland and organic rich soils are estimated to store 1,620 megatonnes of carbon, such storage contributing to climate regulating services and placing Scottish soils in a central role in the delivery of a low carbon economy through C sequestration. The key Scottish Government policy in relation to soils and a low carbon economy is the Soils Framework (The Scottish Government 2009a). As noted in the framework document, the slow rate of soil formation (estimated as 0.04–0.08 mm per year for mineral soils (<1 cm per century) and peat formation (0.8 mm/yr, which is equivalent to a carbon accumulation rate of 0.5 tonnes carbon/hectare/year in actively growing bogs of 'good' habitat status) is not matched by its potential for reversal i.e. "soils can be quickly degraded and lost." Consequently, existing policies and strategies fully recognise the need to protect land that has carbon-rich soils, such as peatlands and upland organo-mineral soils.

Sympathetic land-use and management of the soil, including restoration of peatlands, has the potential to reduce CO_2 emissions by hundreds of thousands of tonnes (Royal Society of Edinburgh 2011). Currently, farmers are incentivised to protect soil carbon via the Single Farm Payment scheme under which 'good agricultural and environmental condition' (GAEC) must be maintained. However, the carbon status of agricultural soils in Scotland is unknown as farmers rarely measure soil carbon, despite its central role in maintaining fertile soils, and there is a lack of quantitative data on the relative merits of different cultivation practices on soil carbon. The total soil-carbon sequestration potential for the UK has been variously estimated at 30–70Mt yr^{-1}. However, Smith *et al* (2005) showed that the achievable level of soil-carbon sequestration is far less than the biophysical potential, depending upon a range of economic, social, institutional and other barriers, estimated to be around one-third to one-half of the biophysical potential (Smith et al., 2007). A preliminary assessment of the amount of carbon losses and gains in Scottish soils using a simple budgeting approach and applied to the Scottish Soils Knowledge and Information Base, shows that either 122Mt or 213Mt of carbon could be lost or gained respectively from the top 30 cm of cultivated mineral soils based on observed minimum and maximum soil carbon values for Scottish soils. But even if 1% of either value were achievable in 40 years, it would be a significant amount relative to net emissions.

2.3.4 Crops

Crops provide us with the 'goods' of food and fibre, and are considered as a provisioning ecosystem service. In the UK, provision of food through plant cropping has increased through land use change, technological improvements and system changes. Between 1940 and 2009, the area of cropped land increased by *c.* 30% in the UK (but a 20% decline in Scotland), with more than 50% of this area under cereals such as wheat. Soft fruit cropping area has declined since 1940, although the proportion of soft fruit crop area grown under protection has increased dramatically in the last 10-15 years. A four-fold increase in agricultural productivity, driven by introduction of high-yielding varieties of crops, better pest and disease control and more effective nitrogen use, has enabled UK self-sufficiency to increase from 40% in the 1940s to 55% in the 2000s, despite a 30% increase in population size. Cereal yields have increased steadily since the 1940s, but have been fairly stable since the mid-1990s at *c.* 8 t/ha for spring wheat and 6 t/ha for spring barley, oats and rye. Cereal production in the UK is presently worth more than £2.5 billion, although the contribution of food production to the UK GDP has fallen from 3% in 1973 to 0.6% in 2009.

Crops are likely to play a major role in delivery of a low carbon economy, and the crop biofuel 'goods' of fibre, bioethanol and biodiesel, along with more general food and fibre 'goods' produced by reduced input farming are most relevant to this policy goal. Land used for crops might be diverted into alternative land uses that support renewable energy production (e.g. biocells, forestry) rather than food production, leading to potential conflict with crop 'good' provision. Alternatively the type of crops grown on cropped land might change to either accommodate increased demand for particular energy crops (biomass, bioethanol), or to enable particular farming practices to be implemented in order to reduce emissions during crop production. The influence of this ecosystem service on delivery of a low carbon economy will depend to a large extent on its relative value for food production *versus* that for energy production, with potential conflict between the crop 'goods' of food and energy for the delivery of a low carbon economy. In particular, the process of producing annual crops for food or fuel requires high levels of fertiliser use and farming practices such as annual ploughing that consume fossil fuels and release carbon into the atmosphere, which would be considered "bad" or "negative good" with respect to a low carbon economy. A recent review in the food *vs.* fuel debate concluded that growing deep-rooted perennial energy crops, preferably on marginal or degraded agricultural lands that are uneconomic for annual food crops, would represent the best opportunity for reducing carbon emissions and resolving land use conflicts for production of food and biofuel crops (Valentine *et al.* 2012). Thus, much of the current Scottish government policy with respect to crops and a low carbon economy is about the methods used to deliver this ecosystem service (optimal crop genotypes, fertiliser use, farming practices etc), although this might incidentally result in greater reliance on non-crop biodiversity for delivery of 'goods' from low input crops.

2.4 How do biodiversity and biotic and biophysical processes underpin these services and goods?

2.4.1 Trees

Biophysical processes such as rock and soil type will influence woodland characteristics (UK NEA Ch. 8), this will in turn influence the ecosystem services provided by the trees. Different tree species will grow in different biophysical conditions (Paterson 1994). Indeed, in some biophysical conditions (extreme cold, drought, wind, heat) trees will not grow at all. Biophysical conditions will therefore influence which tree species (if any) will grow, and how well they will grow. The effects of biotic processes, as discussed below, are then imposed on the distribution of tree species as controlled by abiotic environmental conditions, and it is a combination of abiotic and biotic factors together with forestry management practices that will influence the quality or amount of ecosystem services delivered. Although forestry management practices are outside the remit of this work, in forestry situations, as opposed to unmanaged woodlands, management practices are a major driver of total carbon storage in trees (Morison *et al.* 2012).

The Ecological Site Classification (ESC) (Pyatt *et al.* 2001) and the Native Woodland Model (NWM) (Towers *et al.* 2004) are both models that allow a strategic level of prediction of the most suitable areas for woodland establishment based on biophysical details. The ESC cover the whole of the UK and is designed to match key site factors (elevation, climate, windiness and soil type) with the ecological requirements of different tree species and woodland communities, as defined in the National Vegetation Classification (Rodwell 1991) for Great Britain. The NWM covers Scotland and is based on the 1:250 000 scale National Soils Map, the 1:25 000 Scale Land Cover Map of Scotland 1988, knowledge of the requirements of different types of native woodlands in Scotland using Rodwell & Paterson (1994), a range of other published sources (see Towers *et al.* 2004) and expert knowledge. It shows the predicted potential extent of native woodland cover under current biophysical conditions. Both the ESC and the NWM models illustrate the linkages between biophysical conditions and potential tree cover (and hence the ecosystem services provided by trees). In plantation woodlands, where C storage may be greatest, the limitations imposed by biophysical conditions may to some extent be overridden by management practices such as fertilisation and drainage. However in a low carbon economy such practices generally result in a negative impact on service delivery through the use of fossil fuels and the release of CO_2. Thus, practices to overcome the limitations imposed by biophysical and biotic processes, and thus deliver more tree-sequestered carbon, may ultimately have to be traded-off with the overall goal of delivering a low carbon economy.

Climate change will influence where trees are able to grow and which species may be present. Significant changes in tree species composition and distribution have been found in parts of continental Europe following increased temperatures (e.g. Jump *et al.* 2006a; Penuelas *et al.* 2007). This is most noticeable at the range edges of a species, for example increased reproduction and growth has been detected at the northern limit of *Pinus sylvestris* as a response to increased temperature, whereas at its southern limit increased drought stress has resulted in decreased growth (Matias & Jump 2012). Such changes will influence both the amount of carbon stored (due to changes in tree growth rates and forest composition) and where the carbon is stored (e.g. increases in the altitudinal range of some tree species).

While climate will influence where different species of trees will grow, extreme climatic events can result in an abrupt decline in carbon sequestration, sometimes with long-term impacts. For example the effects of the 1976 drought on the beech (*Fagus sylvatica* L.) in Lady Park Wood in Wales were still visible 15 years after the drought. Damaged trees were still dying 15 years later from drought-induced damage, and the growth of survivors was still negligible until about 1985 and never recovered to pre-drought rates (Peterken & Mountford 1996). Studies in France have also found long-term impacts of extreme climatic events on tree growth and hence carbon sequestration (Breda & Badeau 2008). With extreme climatic events being predicted to increase with climate change (IPCC 2012), this could lead to a significant degradation in the regulating service provided by trees in some areas.

Biotic processes can enhance or reduce tree growth and therefore influence the services delivered by trees. Terrestrial micro-organisms, non-lichenized fungi, lichens, bryophytes and land plants are all listed by the UK NEA (Ch. 4) as of high importance in the delivery of the ecosystem services provided by trees/standing vegetation. However the UK NEA does not separate out the various roles of biodiversity between trees/standing vegetation and peat, for instance other literature shows us that bryophytes are more important in peat formation than trees.

In terms of the positive effects of biotic processes, many tree species have mycorrhizal associations which are important for tree establishment and obtaining nutrients, and tree growth may be severely reduced if mycorrhizal associations are absent (e.g. James *et al.* 1978; Bardgett & Wardle 2010; Collier *et al.* 2012). In addition, a substantial proportion of the carbon fixed by the trees is allocated below ground to ectomycorrhizal (EM) symbionts. Mycorrhizae are therefore important indirectly for carbon sequestration by increasing tree growth, but also directly by increasing the amount of carbon stored in the mycorrhizae themselves (Wallander et al 2011). The mycorrhizal associations will change with changes in environmental conditions e.g. drought or pollution (Erland & Taylor 2002; Bingham & Simard 2012). In addition, insects, birds and mammals are all important for pollination and dispersal of tree seeds/fruits. The importance of these groups depends on the tree type and the surrounding landscape (Breitbach *et al.* 2012). However, this interaction with biotic processes for dispersal is only of importance when considering the natural spread of trees rather than the establishment of trees through planting. In terms of carbon sequestration, biotic interactions that influence tree survival and growth after establishment, e.g. pests and diseases, are the key ones to consider.

Heavy grazing by deer and sheep can negatively influence woodland regeneration and tree growth (Palmer *et al.* 2003a, b, 2004; Gill 2006) and hence reduce the regulating service provided by trees. However, the UK NEA (Ch. 15) states that the most significant threat to the future supply of provisioning and regulating services from woodlands comes from pests and diseases. For example, over the course of the last century, Dutch Elm Disease (caused by the fungus *Ophiostoma novo-ulmi* and spread by the beetles of *Scolytus* species) has severely reduced the abundance and geographic distribution of elms in the UK. Another example is Sudden Oak Death, where the fungus *Phytophthora ramorum* can cause the sudden death of both UK native oak species. The last decade has also witnessed a gradual increase in oak mortality attributed to Acute Oak Decline which is thought to be caused by multiple agents, including various species of bacteria (Brady *et al.* 2010, Forestry Commission 2010). Similarly, Red Band Needle Blight (caused by the fungus *Dothistroma septosporum*) has, over the last couple of decades, dramatically reduced timber yields in infected plantation pine forests (Forestry Commission 2008). Finally ash dieback *Chalara fraxinea* has been recently widely detected within the UK, with potential for substantial impacts on our native ash population (Defra 2012). With a changing climate there remains the potential for other introduced pests and diseases to have similar or even greater impacts in the future (UK NEA). Biotic processes, and in particular the promotion of pest regulation, may therefore be critical for tree populations and C storage. However, such interactions may be complex and species specific. For instance, exclusion of bird predation led to a significant increase in insect herbivory in a study of woodlands in south-western France but only for *Betula pendula* (birch) not *Quercus robur* or *Q. ilex* (two oak species) (Giffard *et al.* 2012). Several studies have found that seedlings or saplings grown in mixed stands, rather than in pure stands of the same species, have lower insect damage (Batzer *et al.* 1987; Keenan *et al.* 1995; Giffard *et al.* 2012), probably because specialised herbivores are less likely to find a host plant in mixed stands than in pure stands.

Genetic diversity within a tree species is important for disease resistance: generally the more disease resistant a tree is the bigger it will grow and the more carbon it will sequester. Genetic diversity has been shown to be important, for example, for increasing resistance against sudden oak death (Hayden *et al.* 2011); and the huge genetic diversity in willows (*Salix* spp.) is seen as a control measure for the potential threat of rust (*Melampsora epitea* and *M. capraearum*) on short rotation coppice willow (Pei &Hunter 2000). The genetic diversity of Scots Pine (*Pinus sylvestris*) has been shown to be related to diversity of plant secondary metabolites which is related to the resistance of the trees to browsing (Iason *et al.* 2011); this in turn is likely to be related to carbon sequestration with bigger, unbrowsed trees sequestering more carbon.

Genetic diversity within a tree species trees is also important for adaptation to climate change (Jump *et al.* 2006b) and the Forestry Commission is recommending that genetic diversity be conserved to enhance resilience to climate change (Duncan 2008). New species or genotypes may become more suitable for plantation forestry as the climate changes, for example the Forestry Commission has recommended a review of the Douglas fir provenance used in Scotland, as better-suited material may currently be available from more southerly latitudes (Duncan 2008).

Trees are typically thought of as the ecosystem dominant species within a wooded environment, and hence the species that drive the changes in the rest of the plant community (Iason *et al.* 2005; Mitchell *et al.* 2007). However, the ground flora has been shown to strongly influence the survival and growth of young trees, both positively and negatively, for example through protection from herbivores, attracting herbivores, chemical inhibition of growth, and N mineralization (Handley 1961; Mallik 1995; Brooker *et al.* 2006; Millett *et al.* 2006, 2008; Mitchell 2012). Thus habitat diversity, especially the surrounding ground flora, will influence tree growth, particularly in young trees and hence affect delivery of the ecosystem services from trees.

The traits of the trees will influence the amount of carbon sequestered (De Deyn *et al*. 2008). High litter quality implies higher carbon mineralisation and lower carbon sequestration (Mascha *et al.* 2010; Pakeman *et al*. 2011). In temperate forests, traits such as low litter C:N ratio can be important for longer-term soil carbon sequestration through enhanced soil aggregation, litter distribution and nutrient mineralization (Lavelle *et al.* 1997). Fast growth rates will also lead to increased carbon storage. However the total carbon sequestered is not necessarily simply correlated with the community weighted means (CWM – see Ch. 1 for a definition) of these traits. While high rates of decomposition imply high loss of carbon, the net effect on carbon sequestration depends on the efficiency of nutrient mineralization and the re-use in primary production. In addition, the net effect of litter quality on the carbon pool will depend on the traits of the soil fauna and their influence over decomposition rates (De Deyn *et al.* 2008). Trees with mycorrhizal fungi associations lose assimilated carbon through the rapid movement of carbon from photosynthate to soil respiration. However the amount lost through this route varies between tree species for reasons that currently are unknown (De Deyn *et al.* 2008). The efficiency with which carbon is sequestered also depends strongly on forest age- structure, and tree height has been suggested as an important trait for soil carbon sequestration (De Deyn *et al.* 2008). Carbon loss in temperate forests is mainly due to decomposition and major disturbances (e.g. storms, disease). Traits for enhanced resistance and resilience to these disturbance factors (e.g. seed and seedling longevity, tolerance to shade, herbivores, and pathogens) may thus yield enhanced carbon sequestration in the longer term (De Deyn *et al.* 2008). Importantly, the relative importance of the traits discussed above for carbon sequestration is unknown; further work is required to study the scaling up of these traits in relation to carbon storage over large areas, and on the trade-offs between the different traits in relation to carbon storage (cf Lavorel, *et al.* 2012).

2.4.2 Peat

Peat formation results from substantially higher rates of primary production than decomposition, and occurs under biophysical conditions that favour this imbalance (Clymo *et al.* 1998). The ecosystem service of peat formation is therefore regulated by biophysical environmental conditions, and underpinned by the biotic processes of the peat forming plant- and associated litter-decomposing microbial communities which are restricted by these biophysical conditions. The NEA acknowledges that changes in peatland vegetation composition affect carbon cycling through differential rates of assimilation and transfer of recent photosynthetic carbon to soils. By influencing rates of litter decomposition and soil respiration, changes in peatland vegetation composition are also likely to affect peat formation (UK NEA Ch. 5). The UK NEA also states that there is still great uncertainty over the scale of impacts on peat ecosystem service delivery from management practices such as burning and drainage. Loss of plant cover in bogs through sheep grazing and rotational moorland burning for grouse both inhibit peat production, and can expose peat soils which may result in peat erosion (Stevenson *et al.* 1990; Yeloff *et al.* 2006) as well as accelerated biological decomposition. However, a recent study by Cummins *et al.* (2011) failed to identify significant relationships between the area of eroded peatland vegetation across Scotland and the densities of large herbivores based on remote sensing, and analysis of various drivers, including climate, geography and the densities of sheep and red deer. There was also little evidence for a correlation with burning or recreational activity.

For peat formation, the interactions between the physical environment and biotic processes are well established. At a localised scale the influence of both temperature and water table on the balance of carbon fixation and release is regulated by abiotic conditions, but mediated by biotic processes. Turetsky *et al.* (2008) suggest significant differences in primary production and decomposition rates among species and their microtopographic niche (hummock vs. hollow species). However, given that little is known about species, population or individual variation (i.e. phenotypic plasticity) in moss traits, further research is needed to explore relationships between the genetic diversity of mosses, their key traits and environmental parameters (Turetsky *et al.* 2008).

Decomposition of *Sphagnum* litter occurs at a relatively high rate in the oxygenated surface peat horizon (acrotelm) and at a very low rate in the permanently waterlogged anaerobic subsurface (catotelm) (Clymo 1984). Reasons for the low decay rates of *Sphagnum* litter are the acidic, nutrient-poor (particularly in N and P), anaerobic conditions and the presence of various phenolic compounds and waxes, which are highly decay-resistant (Verhoeven & Liefveld, 1997). Under scenarios where production and decomposition have the same sensitivity to temperature, peatland models report an overall domineering effect of increasing temperatures on productivity over decomposition, and thus enhanced peat accumulation (Loisel *et al.* 2012; Frolking *et al.* 2001). However, experimental studies have demonstrated a significant response of both short-term (plant-related) and longer-term (peat soil-related) carbon respiration processes to increasing temperatures, with a major proportion of the increase in respiration rate originating from carbon in deeper peat (Dorrepaal *et al.* 2009). The authors conclude that climate warming accelerates respiration of the extensive, subsurface carbon reservoirs in peatlands to a much larger extent than was previously thought. Additionally, the potential effect of increased decomposition at elevated temperatures is exacerbated in combination with atmospheric N deposition. Several studies have demonstrated the detrimental effect of increasing N deposition on growth of ombrotrophic *Sphagnum* species and communities, leading to a reduction in *Sphagnum* growth and a decrease in carbon sequestration (Press *et al.* 1986; Van der Heijden *et al.* 2000; Berendse *et al.* 2001). Recent studies additionally suggest that with decreasing *Sphagnum* primary production at elevated N concentrations, litter decomposition also increases (Pankratov *et al.* 2011; Bragazza *et al.* 2012) exacerbating the negative effect on peat formation and carbon sequestration.

At a broad geographic scale, peat formation is influenced by topography, precipitation, temperature and photosynthetically active radiation integrated over the growing season (PAR_0). Since *Sphagnum*-dominated bryophyte communities are the main peat builders in most high-latitude peatlands, *Sphagnum* productivity and decomposition rates are important for evaluating changes in peat formation and carbon sequestration. A meta-analysis by Loisel *et al.* (2012) concluded that high latitude patterns of peatland *Sphagnum* species growth are driven by PAR_0, and not by moisture. The authors suggest that although considerable uncertainty remains over the carbon balance of peatlands under a changing climate, increasing PAR_0 as a result of lengthening growing seasons due to global warming could promote *Sphagnum* growth if there is no moisture stress, leading to greater peat carbon sequestration and thus peat accumulation. Clark et al. (2010) examined the current topographic and climatic conditions in the areas where upland peat occurs in the UK using bioclimatic envelope models. This study noted that models that included measures of both hydrological conditions (precipitation, potential evapotranspiration and derived variables) and maximum temperature provided the best fit to the mapped peat area. Under the most recent climate change projections, 7 out of the 8 models showed a decline in the bioclimatic space associated with blanket peat. Eastern regions were shown to be more vulnerable than higher-altitude, western areas. A study by Bragazza (2008) investigated the effect of the summer 2003 climate anomaly on *Sphagnum* survival in alpine peatlands. The author established a simple climatic threshold climatic threshold, defined by the ratio of precipitation to temperature (P:T) which triggered the irreversible desiccation of peat mosses. According to Kreyling et al. (2011) extreme climatic events promote stochastic effects in community trajectories and impair predictability even under homogeneous abiotic conditions. However, Smart *et al.* (2010) predicted only small (<1%; between 2020 and 2050) decreases in ombrotrophic *Sphagnum* cover in the UK, using *Sphagnum* cover data from the Countryside Survey of Great Britain, atmospheric pollution data, and climate change predictions in a generalised additive mixed model. The authors also concluded, however, that the uncertainties surrounding this estimate are substantial, and also that the additional uncertainties associated with prediction of atmospheric deposition and soil bio- geochemical processes should be evaluated.

These examples demonstrate uncertainty with respect to the influence of associated decomposer biodiversity on peat assimilation.

2.4.3 Soil formation

The majority of soil processes in both upland and lowland systems, including many of those involved in soil formation, are mediated by the activity of the soil microbial community. Soil biota are present in large numbers and at generally high levels of diversity.

It has been found repeatedly that changes in total microbial biomass and community structure are associated with changes in soil fertility. High fertility conditions reduce the total microbial biomass and shift the dominance towards the bacterial community of the soil, supporting greater net primary productivity and C storage through increased plant growth. However, with closed nutrient cycling, i.e. in less fertile soils, fungal food webs tend to dominate, with slower plant growth and reduced productivity (Bardgett *et al.* 1996; Grayston *et al.* 2004; Van der Heijden *et al.* 2008). In nutrient poor, acidic upland grassland soils, slow nitrification and mineralization can often result in the availability of nitrogen limiting productivity. In such soils, nutrient availability is heavily dependent on microbial activity and in such cases up to 20% of nitrogen and 75% of phosphorous can be provided by nitrogen-fixing bacteria and mycorrhizal fungi (Van der Heijden *et al.* 2008).

Studies on N-limited grasslands have shown that while high diversity mixtures of grassland species stored five-fold more soil carbon than did monoculture plots, the joint presence of grasses and legumes on such N-limited soils was a key cause of greater soil C (and N) accumulation regardless of plant assemblage diversity. This was due to the transfer of fixed N via the nitrogen-fixing bacteria/legume symbiosis into increased below-ground biomass and thus soil C and N inputs from the associated grasses (Fornara and Tilman, 2008). Whether or not high levels of diversity below-ground are necessary for the retention and maintenance of desired functions, e.g. those leading to increased carbon sequestration, has been argued over at length, with varying but significant levels of functional redundancy found within different processes (reviewed in Nielsen *et al.* 2011). Thus, it can be argued that a reduction or loss in key functional diversity in either above- or below-ground species could be critical to the balance of such systems and their ability to effectively sequester carbon.

Changes in plant community structure as a consequence of pasture improvement or cultivation also affect microbial community structure and diversity (Grayston *et al.* 2001, 2004; Buckley & Schmidt 2003; McCulley & Burke 2004), and increases in microbial diversity are generally associated with more diverse plant communities (Kowalchuk *et al.* 2002). However, the impact of this change on soil formation and ultimately C storage is unclear. An intensive study at Sourhope, a Scottish upland grassland site, investigated the links between function and soil biodiversity (Usher *et al.* 2006). Despite a lack of diversity in the vegetation at Sourhope, it was shown that the site's soil was extremely bio-diverse, and a soil food web including carbon flow allowed for the clear demonstration of the importance of the movement of carbon from plants into soil fungi and onwards to bacterial communities (Leake *et al.* 2006; Krsek *et al.* 2006) and thus the importance of soil microbial populations to ecosystem form, function and eventually services.

Understanding the effects of soil management processes on micro- as a well as macro-biota is critical to enabling sustainable management practices (Irvine *et al.* 2007). For example, nitrogen and lime fertiliser treatment was shown to reduce soil moisture content and this was attributed to the increase in above-ground biomass, causing an increase in evapo-transpiration. This indicated that changes in soil moisture content could 'over-ride' plant influences in microbial population regulation, and that the same may well be true for nematodes and other water-film reliant micro-fauna (Murray *et al.* 2006). The significant contributions of these soil organisms in the formation and stabilisation of soil structure is widely documented, but only induced changes in soil fauna following the disturbance of natural soil structure were found to show significant shifts in community structure, leading to the conclusion that although the activities of soil fauna largely create the structure of such grasslands, relatively little of that structure may be biologically active with respect to current processes in undisturbed environments.

This supports the finding that it is largely only changes in key species or communities with strong ecological functions that significantly impact soil processes, and that the huge diversity sometimes found can be largely redundant (Davidson & Grieve 2006). That said, the Sourhope soil ecosystem was highly stress resistant due to a combination of taxonomic diversity and rapid carbon flux (Fitter *et al.* 2005). Consideration of key drivers rather than diversity overall would therefore be indicated when management is considered. Whether this can be applied to other soil types, and to soil formation as a whole, is largely undetermined. Further, completely discounting currently 'inactive' diversity would appear unwise without a full understanding of the soil environment and its processes, and this remains something that is lacking (Havlicek 2012).

2.4.4 Crops

We have combined information concerning the generic role of biodiversity in supporting crop production into the Sustaining Food Production chapter of this review (Ch. 3). Here, we focus instead on the roles of biodiversity and biophysical processes in supporting those 'goods' from the ecosystem service of crops that are specific to delivery of a low carbon economy, namely crop biofuel 'goods' of fibre, bioethanol and biodiesel, along with more general food and fibre 'goods' produced by reduced input farming. However, it should be noted that the contribution of the ecosystem service of crops to a low carbon economy can depend on the specific crop type. Some crops particularly relevant to delivering a low carbon economy (e.g. short rotation coppice or *Miscanthus*) are currently being developed, but their relationship to underlying biodiversity and biophysical processes (and, indeed, their impact on biodiversity) is still unclear. For now, we can only base our assessment on the known relationships between biodiversity, biophysical processes, and well-studied crop-types.

As for trees, the abiotic environment and its associated impact on soil conditions set absolute limitations on crop production within which the influence of biotic processes can then operate. As discussed in more detail in Section 3, the current agricultural practice for crop production in Scotland means that there is a relatively limited dependence of the production of some crops on biotic processes: the services that might be provided by biotic processes are instead provided artificially (e.g. through nutrient, pesticide or herbicide applications). However, a shift toward a low carbon economy (either actively, or in response to increased fuel costs), might necessitate a reduction in agricultural inputs and a greater dependency on biotic processes for the provision of key functions.

This, in turn, will influence the desirable crop traits that are required to maximise food production. For example, high grain yield and nitrogen use efficiency in modern barley varieties have been achieved through selection of desirable traits in high mineral nitrogen growing environments (Bingham *et al.* 2012); crop productivity in reduced input systems, with relatively greater reliance on organic nitrogen sources, is likely to require alternative crop traits to maximise yield (e.g. Pswarayi *et al.* 2008).

Decreased carbon emissions from crop production systems could be achieved by maximising carbon storage in agricultural soils, which can be achieved by breeding crops with below-ground C sequestration traits such as increased root size (Kell 2012), or increasing the relative production of winter annuals, biennial or perennial crops with larger or more persistent root systems. Alternatively, recycling of non-food residues into fuel production could mitigate carbon emissions associated with annual crop production. However, Valentine *et al.* (2012) argue that incorporation of crop residues into soil would be a more effective and sustainable approach to reducing carbon emissions due to the transport and infrastructure costs associated with using low bulk density residues for energy production.

Making better use of biotic processes to support crop production is already seen as a vital element of developing more sustainable (including less energy-intensive) farming systems. For example, understanding the contribution of biotic processes to crop production is a key target of integrated pest management strategies, driven by the fact that insect pests are estimated to destroy 30-40% of crop yield pre- and post-harvest, an amount that could feed one billion people, despite an estimated US$33.19 billion spent globally each year on pesticides (Birch *et al.* 2011). Increased diversity of herbivore natural enemies, promoted by increased diversity of functional traits of vegetation (e.g. Wäckers 2004) in the vicinity of crops (for example by the provision of buffer strips) could reduce pest impacts in crops (Jonsson *et al.* 2008). Similarly, increased genetic diversity within crop species can prevent the spread of disease and reduce the need for pesticide applications (Newton *et al.* 2009) although the specific plant traits underlying these effects are not always clear.

Weeds can be significant components of the vegetation biomass and diversity in agricultural systems, but, as crop productivity often declines at high weed densities (Karley *et al.* 2011; Hoad *et al.* 2008), weeds are usually maintained at low abundance and diversity. However, weed vegetation supports a greater diversity and abundance of insects than crop vegetation (Karley *et al.* 2011), and moderate increases in weed abundance and diversity could have positive effects that ultimately help deliver crop production with reduced levels of artificial inputs and energy expenditure. These include increased abundance and diversity of insect pollinators (Gibson *et al.* 2006), reduced prevalence of pests and disease (Iannetta *et al.* 2010), and capture of nutrients that might otherwise be leached from arable soils (Karley *et al.* 2010). Similarly, in lowland grassland systems, increased vegetation biodiversity can have a positive effect on productivity and N uptake (Bessler *et al.* 2012). Overall, however, changes in crop production that result from variation in microbial, invertebrate and vegetation diversity are relatively small compared to the response of crop production to variation inputs from intensive management practice, and geographic and temporal variation in soil abiotic factors (e.g. Seufert *et al.* 2012).

2.5 Interactions between Ecosystem Services

Within the ecosystem services prioritised for a low carbon economy, the most common trade-offs occur around land use and land management. Decisions have to be made over which crops (food or biofuels) or habitats (woodland or other habitats) to have on any given piece of land, with an acknowledgement that these decisions will influence the success of policies to implement a low carbon economy. This influence will be due to impacts on either: a) the potential for soil or vegetation to sequester carbon, or: b) carbon emissions from related land management practices. Ecosystem services mapping is one way to assess trade-offs in how the land may be used, allowing spatially explicit strategic visioning and decision-making (Brown 2011). This tool also allows the impacts of climate change and changes in the trade-offs between different land-uses to be assessed (e.g. Gimona *et al.* 2012).

Clear trade-offs exist between the desire for increased woodland cover (Scottish Executive 2006) and the use of land for other purposes that help to sequester carbon, e.g. crops or moorland. During 2005 to 2009 between 3440 and 6594 ha of new woodlands were planted each year in Scotland, with an annual net carbon sequestration of between 0.36 MtCO2 and 0.43 MtCO2 (The Scottish Government 2010a). Increased woodland cover on organic-rich soils may lead to a decline in the carbon stored in the soil (due to increased decomposition rates), which is not balanced by the increase in the carbon sequestered by the trees (Mitchell *et al.* 2007; Scottish Executive Environment and Rural Affairs Department 2007). Carbon stocks held in trees are generally dwarfed by those held in the soil (Scottish Executive Environment and Rural Affairs Department 2007). Changes in soil properties and soil function due to woodland expansion should be taken into account when calculating the overall impact of tree establishment on carbon storage. In addition, management practices promoting tree growth (e.g. fertiliser addition) will have negative consequences for achieving a low carbon economy. The planned expansion of woodlands in Scotland will cause the loss of other semi-natural habitats, e.g. upland grasslands, moorlands or agricultural land (Woodland Expansion Advisory Group 2012). Much of the land identified for woodland expansion is farmland, in particular grazing land. There will therefore also be a trade-off between woodland creation and livestock farming (Woodland Expansion Advisory Group 2012).

There are clear potential trade-offs between crop production and soil formation. Soil cultivation practices for crop production are typically thought to degrade soil structure, density and composition and to promote soil loss through erosion (Morris *et al.* 2010; Powlson *et al.* 2011). Economic pressures in recent years have encouraged the development of tillage systems that minimise the costs associated with soil cultivation (Morris *et al.* 2010). Non-inversion tillage minimises disturbance of the soil and reduces the risk of developing compacted soils that mechanically impede plant root development. Non-inversion tillage also allows plant residues to accumulate at the soil surface, which improves topsoil structure and stability and thus reduces risk of soil erosion (Morris *et al.* 2010). However, the apparent negative impact of conventional soil tillage on soil organic carbon sequestration and soil microbial community structure might actually result from changes in the distribution with depth, rather than loss, of soil carbon and microbial biomass (Sun *et al*. 2010), and further research is needed to determine the generality of these findings.

Reduced tillage systems have been claimed to reduce soil organic carbon (SOC) decomposition and so increase SOC sequestration, but much of this effect has recently been shown to be due to the concentrating effect of SOC near the soil surface (Baker *et al.* 2007; Angers & Eriksen-Hamel 2008). That said, such approaches can have numerous beneficial effects including decreased erosion rates and improved soil structure in many soil types and crop systems. However, in relation to climate change mitigation, these benefits could be outweighed by increased emissions of nitrous oxide, especially in moist environments (Baggs *et al.* 2003; Ball *et al.* 2008; Rochette 2008) resulting in a conflict between soil management to conserve soil C versus promoting emission of GHGs from soil. The use of locally relevant data coupled with new research is required to assess whether reduced tillage systems can be applied effectively in Scotland as part of a strategy for achieving a low carbon economy.

For soils already under crops, opportunities for additional carbon storage are limited when changes in land management due to simple transfer of organic carbon (e.g. application of manure) from one location to another are considered in their true light (Powlson *et al.* 2010). For an increase in soil organic carbon to positively affect climate change there must be a net transfer of C from the atmosphere to the soil, due to additional photosynthesis and nutrient transfer, or through a reduced rate of soil organic matter (SOM) decomposition. Thus, major changes in management practices are often necessary before genuine sequestration of additional carbon to agricultural soils is achieved. Possible approaches such as use of crops with deeper root systems, intercropping, or agroforestry may achieve this, but in many cases more research is required (Powlson *et al*. 2011).

Trade-offs will also have to be made between the 'goods' and services associated with delivering a low carbon economy and those associated with delivering other broad policy goals. Within the broader ecosystem service of crops there are conflicts over which types of crops to grow: food or biofuels. Biofuels may reduce the use of fossil fuels, but food and fibre are important to support a growing population, and locally-grown food is also part of a low carbon economy. As noted, land use change from natural vegetation to crop growth will generally result in a decline in soil organic carbon content. This then also has knock-on effects for other ecosystem services: even small changes in soil carbon can have large negative effects on other ecosystem services such as water storage and soil formation (Blair *et al.* 2006; Watts *et al.* 2006). However, some clear synergies also exist. Peat formation is clearly essential for C sequestration, and this needs to be associated with particular habitats which are also commonly of high importance for biodiversity conservation.

2.6 Knowledge gaps

With respect to trees, more detailed measurement and development of associated predictive models are needed with respect to the changes in soil C following afforestation.

With respect to peat, there is considerable uncertainty over the rates of peat formation and loss, as well as over the scale of impacts on peat development and maintenance of management practices such as heather burning and drainage. Further research is also needed to explore relationships between key moss traits and environmental parameters and consequent ecosystem processes, whilst comprehensive studies of the relationships between external drivers and the net primary production and decomposition of peat are currently lacking. Trends in potential external drivers can be seen. For example, Countryside Survey data between 1998 and 2007 indicate a small decline in mean plant species richness in Scottish Bogs, and increases in the proportion of competitive species relative to ruderal species (UK NEA Ch. 5). Therefore, biodiversity changes seem to be evident in a Scottish context, but the impacts on peat formation are not clear. As noted above, there are also clear uncertainties about the relationships between atmospheric N pollution and *Sphagnum* litter decomposition rates in acrothelm and deeper catothelm layers.

With respect to soil formation, the carbon status of agricultural soils in many areas of Scotland is unknown as farmers rarely measure soil carbon, despite its central role in maintaining fertile, healthy and resilient soils. Consequently our understanding of the biophysical/biotic process regulation of carbon fixation is limited. Although data exist on soil carbon concentration in arable soils in Eastern and Central Scotland, further work is needed to quantify the relative merits of different cultivation practices for C sequestration in soils. With respect to biotic interactions regulating soil formation, although we know that plants impact on soil organisms, we do not know how this then influences soil formation.

With respect to crop production to achieve a low carbon economy, although there is pressure to reduce inputs for crop production (nutrient and pesticide production and the fuel required to apply them and to cultivate the soil), we do not know enough about the position of the tipping point in the trade-off between reducing inputs and sustaining ecosystem service provision and 'goods' from crops. With respect to new crops such as biofuels, we know very little about their underpinning by, and impacts on, biodiversity and biotic processes. The use of locally relevant data coupled with new research is required to assess whether reduced tillage systems can be applied effectively in Scotland as part of a strategy for achieving a low carbon economy.

Overall there is clearly a lack of quantitative data allowing us to assess the relative importance of biodiversity *per se*, and/or key functional groups, for maintaining function and ecosystem service provision. Furthermore, and assuming that there is a relationship between biodiversity and ecosystem service provision, is there functional redundancy in the provision of 'goods' associated with peat, crops, trees, and soil formation? There is some indication that the huge organism diversity sometimes found in soils can be largely redundant (Davidson & Grieve 2006), but specifics of which species or species groups are redundant and which species are keystone are lacking. It is unknown whether or not there is also functional redundancy for peat, trees and crops.

References

Alexander, P. D., Bragg, N. C., Meade, R., Padelopoulos, G. & Watts, O. (2008) Peat in horticulture and conservation: the UK response to a changing world. *Mires and Peat*, **3**: 1-10.

Angers, D.A. & Eriksen-Hamel, N.S. (2008) FFull-inversion tillage and organic carbon distribution in soil profiles: a meta-analysis**.** *Soil Science Society of America Journal*, **72**, 1370–1374.

Baggs, E.M., Stevenson, M., Pihlatie, M., Regar, A., Cook, H. & Cadisch, G. (2003) Nitrous oxide emissions following application of residues and fertiliser under zero and conventional tillage. *Plant and Soil,* **254**, 361–370.

Baker, J.M., Ochsner, T.E., Venterea, R.T. & Griffis, T.J. (2007) Tillage and soil carbon sequestration – What do we really know? *Agriculture, Ecosystems and Environment*, **118**, 1–5.

Ball, B.C., Crichton, I. & Horgan, G.W. (2008) Dynamics of upward and downward N_2O and CO_2 fluxes in ploughed or no-tilled soils in relation to water-filled pore space, compaction and crop presence. *Soil & Tillage Research*, **101**, 20–30.

Bardgett, R.D. & Wardle, D.A. (2010) *Aboveground-Belowground Linkages: Biotic Interactions, Ecosystem Processes, and Global Change*. Oxford Series in Ecology and Evolution, Oxford University Press, Oxford.

Bardgett, R.D., Hobbs, P.J. & Frostegård, Å. (1996) Changes in soil fungal:bacterial biomass ratios following reductions in the intensity of management of an upland grassland. *Biology and Fertility of Soils*, **22**, 261–264.

Batzer H.O., Benzie J.W. & Popp M.P. (1987) Spruce budworm damage in aspen/balsam fir stands affected by cutting methods. *Northern Journal of Applied Forestry*, **4**, 73–75.

Berendse, F., Van Breemen, N., Rydin, H., Buttler, A., Heijmans, M., Hoosbeek, M.R., *et al.* (2001) Raised atmospheric CO_2 levels and increased N deposition cause shifts in plant species composition and production in *Sphagnum* bogs. *Global Change Biology*, **7**, 591-598.

Bessler H., Oelmann Y., Roscher C., Buchmann N., Scherer-Lorenzen M., Schulze E.-D., *et al.* (2012) Nitrogen uptake by grassland communities: contribution of N_2 fixation, facilitation, complementarity, and species dominance. *Plant & Soil*, **358**, 301-322.

Bingham, I.J., Karley, A.J., White, P.J., Thomas, W.T.B. & Russell, J.R. (2012) Analysis of improvements in nitrogen use efficiency associated with 75 years of barley breeding. *European Journal of Agronomy*, **42**, 49-58.

Bingham, M.A. & Simard, S.W. (2012) Mycorrhizal networks affect ectomycorrhizal fungal community similarity between conspecific trees and seedlings. *Mycorrhiza*, **22**, 317-326

Birch A.N.E., Begg, G.S. & Squire, G.R. (2011) How agro-ecological research helps to address food security issues under new IPM and pesticide reduction policies for global crop production systems. *Journal of Experimental Botany*, **62**, 3251-3261.

Blair, N., Faulkner, R.D., Till, A.R. & Poulton, P.R. (2006) Long-term management impacts on soil C, N and physical fertility. Part 1: Broadbalk experiment. *Soil and Tillage Research*, **91**, 30–38.

Bonn, A., Holden, J. Parnell, M., Worrall, F., Chapman, P.J., Evans. C.D., *et al.* (2009). *Ecosystem services of peat – Phase 1 Project code : SP0572*. *Synthesis*. DEFRA commissioned report to Peak District National Park Authority (Moors for the Future), Edale, UK.

Brady, C., Denman, S., Kirk, S., Venter, S., Rodriguez-Palenzuela, P. & Coutinho, T. (2010) Description of *Gibbsiella quercinecans* gen. nov., sp. nov., associated with Acute Oak Decline. *Systematic and Applied Microbiology*, **33**, 444-450

Bragazza, L., Buttler, A., Habermacher, J., Brancaleoni, L., Gerdol, R., Fritze, H., Hanajík, P., *et al.* (2012). High nitrogen deposition alters the decomposition of bog plant litter and reduces carbon accumulation. *Global Change Biology*, **18,** 1163-1172.

Breda, N. & Badeau, V. (2008) Forest tree responses to extreme drought and some biotic events: Towards a selection according to hazard tolerance? *Comptes Rendus Geoscience*, **340,** 651-662.

Breitbach,N., Tillmann,S., Schleuning, M. & Grünewald C. (2012) Influence of habitat complexity and landscape configuration on pollination and seed-dispersal interactions of wild cherry trees. *Oecologia,* **168,** 425 – 437

Brooker, R. W., Scott, D., Palmer, S. C. F. & Swaine, E. (2006) Transient facilitative effects of heather on Scots pine along a grazing disturbance gradient in Scottish moorland*. Journal of Ecology*, **94**, 637-645.

Brown, I., Poggio, L., Gimona, A. & Castellazzi, M. (2011) Climate change, drought risk and land capability for agriculture: implications for land use in Scotland. *Regional Environmental Change*, **11**, 503-518

Buckley, D.H. & Schmidt, T.M. (2003) Diversity and dynamics of microbial communities in soils from agro-ecosystems. *Environmental Microbiology*, **5**, 441–452.

Clark, J., Gallego-Sala, A., Allott, T., Chapman, S., Farewell, T., Freeman, C., House, J., *et al.* (2010) Assessing the vulnerability of blanket peat to climate change using an ensemble of statistical bioclimatic envelope models. *Climate Research*, **45** 131-150.

Clymo, R. S., Turunen, J. & Tolonen, K. (1998) Carbon accumulation in peatland. *Oikos*, **81**, 368-388.

Clymo, R.S. (1984) The limits to peat bog growth. *Philosophical Transactions of the Royal Society of London B-Biological Sciences*, **303**, 605–654.

Collier, F.A. & Bidartondo, M.I. (2009) Waiting for fungi: the ectomycorrhizal invasion of lowland heathlands. *Journal of Ecology*, **97**, 950-963.

Cummins, R., Donnelly, D., Nolan, A., Towers, W., Chapman, S., Grieve, I. *et al.* (2011) *Peat erosion and the management of peatland habitats.* Scottish Natural Heritage Commissioned Report No. 410. SNH, Inverness.

Davidson, D.A. & Grieve, I.C. (2006) Relationships between biodiversity and soil structure and function: evidence from laboratory and field experiments. *Applied Soil Ecology*, **33**, 176–185.

De Deyn, G.B. Cornelissen J.H.C. & Bardgett R.D. (2008) Plant functional traits and soil carbon sequestration in contrasting biomes. *Ecology Letters*, **11**, 516-531.

Defra (2012) *Interim* Chalara *Control Plan*. Defra, London. http://www.defra.gov.uk/publications/files/pb13843-chalara-control-plan-121206.pdf

Dorrepaal, E., Toet, S., van Logtestijn, R. S. P., Swart, E., van de Weg, M. J., Callaghan, T. V., *et al.* (2009) Carbon respiration from subsurface peat accelerated by climate warming in the subarctic. *Nature*, **460***,* 7255.

Duncan, R. (2008) *Impacts of climate change on forests and forestry in Scotland*. Forest Research, Edinburgh.

Erland, S. & Taylor, A.F.S. (2002) Diversity of ecto-mycorrhizal fungal communities in relation to the abiotic environment. *Mycorrhizal Ecology. Ecological Studies Volume 157* (eds M.G.A van der Heijden, I.R. Sanders), pp. 163-200. Springer-Verlag, New York.

Fitter, A.H., Gilligan, C.A., Hollingworth, K., Kleczkowski, A., Twyman, R.M., Pitchford, J.W., *et al.* (2005) Biodiversity and ecosystem function in soil. *Functional Ecology,* **19**, 369–377.

Forestry Commission (2003) *Forests, Carbon and Climate Change: the UK Contribution*. http://www.forestry.gov.uk/PDF/fcin048.pdf/$FILE/fcin048.pdf Forestry Commisssion, Edinburgh.

Forestry Commission (2008) *Red band needle blight of conifers in Britain. Research Note 002*. Forestry Commission, Farnham.

Forestry Commission (2010) *Managing Acute Oak decline, Practice Note 015.* Forestry Commission, Farnham.

Forestry Commission Scotland (2010) *The right tree in the right places. Planning for forestry and woodlands.* Forestry Commission Scotland, Edinburgh.

Fornara, D.A. & Tilman, D. (2008) Plant nfuctional composition influences rates of soil carbon and nitrogen accumulation. *Journal of Ecology*, **96**, 314-322.

Frolking, S., Roulet, N.T., Moore, T.R., Richard, P.J.H., Lavoie, M. & Muller, S. D. (2001). Modeling northern peatland decomposition and peat accumulation. *Ecosystems*, **4**, 479-498.

Gibson R.H., Nelson I.L., Hopkins G.W., Hamlett B.J. & Memmott J. (2006) Pollinator webs, plant communities and the conservation of rare plants: arable weeds as a case study. *Journal of Applied Ecology,* **43**, 246-257

Giffard, B. Corcket, E., Barbaro, L. & Jactel, H. (2012) Bird predation enhances tree seedling resistance to insect herbivores in contrasting forest habitats *Oecologia,* **168,** 415-424

Gill, R. (2006) The influence of large herbivores on tree recruitment and forest dynamics. *Large herbivore ecology, ecosystem dynamics and conservation* (eds K. Danell, R. Befgstrom, P. Duncan, and J. Pastor), pp. 170-202. Cambridge University Press, Cambridge.

Gimona, A., Poggio, L., Brown, I. & Castellazzi, M. (2012) Woodland networks in a changing climate: Threats from land use change. *Biological Conservation*, **149**, 93-102

Grayston S.J., Campbell, C.D., Bardgett, R.D., Mawdsley, J.L., Clegg, C.D., Ritz, K., *et al.* (2004) Assessing shifts in soil microbial community structure across a range of grasslands of differing management intensity using CLPP, PLFA and community DNA techniques. *Applied Soil Ecology,* **25**, 63–84.

Grayston, S.J., Griffith, G.S., Mawdsley, J.L., Campbell, C.D. & Bardgett, R.D. (2001) Accounting for variability in soil microbial communities of temperate upland grassland ecosystems. *Soil Biology and Biochemistry*, **33**, 533–551.

Handley, W.R.C. (1961) Further evidence for the importance of residual leaf protein complexes in litter decomposition and the supply of nitrogen for plant growth. *Plant and Soil*, **15**, 37–73

Havlicek, E. (2012) Soil biodiversity and bioindication: From complex thinking to simple acting. *European Journal of Soil Biology*, **49**, 80-84.

Hayden, K.J., Nettel, A., Dodd, R.S. & Garbelotto, M. (2011) Will all the trees fall? Variable resistance to an introduced forest disease in a highly susceptible host. *Forest Ecology and Management*, **261**, 1781-1791.

Hoad, S., Topp, C., Davis, K. (2008) Selection of cereals for weed suppression in organic agriculture: a method based on cultivar sensitivity to weed growth. *Euphytica,* **163**, 355–366.

Horticultural Trades Association (2011) *Defra Consultation on Peat Reduction: Response from the Horticultural Trades Association*. HTA Press Office, Horticultural House, Reading. http://www.the-hta.org.uk/file.php?fileid=1191

Iannetta, P.P.M., Begg G.S., Valentine, T.A. & Wishart, J. (2010) Sustainable disease control using weeds as indicators: *Capsella* and Tobacco Rattle Virus. *Weed Research***, 50**, 511–514.

Iason, G.R., Lennon, J.J., Pakeman, R.J., Thoss, V. Beaton, J.K., Sim, D.A., *et al.* (2005) Does chemical composition of individual Scots pine trees determine the biodiversity of their associated ground vegetation? *Ecology Letters,* **8**, 364-369.

Iason, G.R., O'Reilly-Wapstra, J.M., Brewer, M.J., Summers, R.W. & Moore, B (2011) Do multiple herbivores maintain chemical diversity of Scots pine monoterpenes? *Philosophical Transactions of the Royal Society B-Biological Sciences*, **366**, 1337-1345.

IPCC (2012) *Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation. A Special Report of Working Groups I and II of the Intergovernmental Panel on Climate Change.* Cambridge University Press, Cambridge, UK, and New York, NY.

Irvine, L., Kleczkowski, A., Lane, A.M.J., Pitchford, J.W., Janvier, C., Villeneuve, F., *et al.* (2007) Soil health through soil disease suppression: Which strategy from descriptors to indicators? *Soil Biology & Biochemistry,* **39**, 1–23.

James, H., Court, M., Macleod, D. & Parsons, J. (1978) Relationships between growth of sitka spruce (*Picea-sitchensis*), soil factors and mycorrhizal activity on basaltic soils in western Scotland. *Forestry*, **51**, 105-119.

Jonsson, M., Wratten, S.D., Landis, D.A. & Gurr, G.M. (2008) Recent advances in conservation biological control of arthropods by arthropods. *Biological Control*, **45**, 172–175.

Jump, A.S., Hunt, J.M. & Penuelas, J. (2007) Climate relationships of growth and establishment across the altitudinal range of *Fagus sylvatica* in the Montseny Mountains, northeast Spain. *Ecoscience*, **14** 507-518

Jump, A.S., Hunt, J.M. & Penuelas, J. (2006a) Rapid climate change-related growth decline at the southern range edge of *Fagus sylvatica. Global Change Biology,* **12,** 2163-2174

Jump, A.S., Hunt, J.M., Martinez-Izquidero, J.A. & Penuelas, J. (2006b) Natural selection and climate change: temperature-linked spatial and temporal trends in gene frequency in *Fagus sylvatica*. *Molecular Ecology*, **15**, 3469-3480

Karley, A.J., Hawes, C., Valentine, T.A., Johnson, S.N., Toorop, P., Squire, G.R., *et al.* (2011) Can arable weeds contribute to ecosystem service provision? Functional diversity in Shepherd's purse (*Capsella bursa-pastoris* L. Medik.). *Aspects of Applied Biology,* **109**, 31-38.

Karley, A.J., Iannetta, P., Valentine, T., Bingham, I., Hoad, S., Young, M., *et al.* (2010) Impact of plant traits on nitrogen-efficiency and compartmentation in arable systems. *Aspects of Applied Biology,* **105**, 89–96.

Keenan, R., Lamb, D. & Sexton, G. (1995) Experience with mixed species rainforest plantations in North Queensland. *wCommonwealth Forestry Review*, **74**, 315–321.

Kell, D.B. (2012) Large-scale sequestration of atmospheric carbon via plant roots in natural and agricultural ecosystems: why and how. *Philosophical Transactions of the Royal Society B-Biological Sciences*, **367**, 1589-1597.

å Kowalchuk, G.A., Buma, D.S., de Boer, W., Klinkhamer, P.G.L. & Van Veen, J.A. (2002) Effects of above ground plant species composition and diversity on the diversity of soil-borne microorganisms. *Antonie Van Leeuwenhoek*, **81**, 509–520.

Krsek, M. & Wellington, E.M.H. (2006) Studies of microbial community structure and function below ground in a managed upland grassland site at Sourhope Research Station. *Applied Soil Ecology,* **33**, 127–136.

Lavelle, P., Bignell, D., Lepage, M. Wolters, V., Roger, P. Ineson, P., *et al.* (1997) Soil function in a changing world: The role of invertebrate ecosystem engineers. *European Journal of Soil Biology*, **33**, 159-193.

Lavorel, S. & Grigulis, K. (2012) How fundamental plant functional trait relationships scale-up to trade-offs and synergies in ecosystem services. *Journal of Ecology*, **100**, 128-140

Leake, J.R., Ostle, N.J., Rangel-Castro, J.I. & Johnson, D. (2006) Carbon fluxes from plants through soil organisms determined by field (CO_2)-C^{-13} pulse-labelling in an upland grassland. *Applied Soil Ecology,* **33**, 152–175.

Loisel, J., Gallego-Sala, a. V. & Yu, Z. (2012). Global-scale pattern of peatland *Sphagnum* growth driven by photosynthetically active radiation and growing season length. *Biogeosciences Discussions*, **9**, 2169-2196.

Mallik, A.U. (1995) Competitive ability and allelopathy of ericaceous plants as potential causes of conifer regeneration failures. *Journal of Korean Forestry Society,* **84**, 394–405.

Mascha J., Karin V., Polle P.A. & Thomas, F.M. (2010) Leaf litter decomposition in temperate deciduous forest stands with a decreasing fraction of beech (*Fagus sylvaticaO*). *Oecologia,* **164**, 1083-1094

Matias, L. & Jump, A. S. (2012) Interactions between growth, demography and biotic interactions in determining species range limits in a warming world: the case of *Pinus sylvestris*. *Forest Ecology and Management*, **282,** 10-22.

McCulley, R.L. & Burke, L.C. (2004) Microbial community composition across the Great Plains: landscape versus regional variability. *Soil Science Society of America Journal*, **68**, 106–115.

Millett, J., Hester, A. J. Millard, P. & McDonald A.J.S. (2006). How do different competing species influence the response of *Betula pubescens* Ehrh. to browsing? *Basic and Applied Ecology,* **7**, 123-132.

Millett, J., Hester, A.J., Millard, P. & McDonald, A. J. S. (2008) Above- and below-ground competition effects of two heathland species: Implications for growth and response to herbivory in birch saplings. *Basic and Applied Ecology*, **9**, 55-66.

Mitchell R.J., Keith A.M., Potts J.M., Ross J., Reid, E. & Dawson L.A. (2012) Overstory and understory vegetation interact to alter soil community composition and activity. *Plant and Soil*, **352**, 65-84.

Mitchell, R.J., Campbell, C.D., Chapman, S.J., Osler, G.H.R., Vanbergen, A.J., Ross, L.R., *et al.* (2007) The cascading effects of birch on heather moorland: a test for the top-down control of an ecosystem engineer. *Journal of Ecology*, **95**, 540–554.

Morison, J., Matthews, R., Miller, G., Perks, M., Randle, T., Vanguelova, E., *et al.* (2012) *Understanding the carbon and greenhouse gas balance of forests in Britain*. Forestry Commission, Edinburgh

Morris N.L., Miller P.C.H., Orson J.H. & Froud-Williams R.J. (2010) The adoption of non-inversion tillage systems in the United Kingdom and the agronomic impact on soil, crops and the environment - A review. *Soil Tillage Research*, **108**, 1-15.

Murray, P.J., Cook, R., Currie, A.F., Dawson, L.A., Gange, A.C., Grayston, S.J., *et al.* (2006) Interactions between fertilizer addition, plants and the soil environment: implications for soil faunal structure and diversity. *Applied Soil Ecology,* **33**, 199–207.

Newton, A.C., Begg G.S. & Swanston, S. (2009) Deployment of diversity for enhanced crop function. *Annals of Applied Biology* **154**, 309–322.

Nielsen, U.N., Ayres, E., Wall, D.H. & Bardgett, R.D. (2011) Soil biodiversity and carbon cycling: a review and synthesis of studies examining diversity-function relationships. *European Journal of Soil Science*, **62**, 105-116.

Paappanen, T., Leinonen, A. & Flyktman, M. (2010) Peat Industry in Finland. *Peat Industry In The Six EU Member States – Country Reports Finland, Ireland, Sweden, Estonia, Lavia, Lithuaniua*. *VTT Research Report VTT-R-06630-10* (ed. T. Paappanen) p. 140. European Peat and Growing Media Association (EPAGMA) commissioned report to VTT Research Technical Research Centre of Finland, Finland.

Pakeman, R.J., Eastwood, A. & Scobie, A. (2011) Leaf dry matter content as a predictor of grassland litter decomposition: a test of the 'mass ratio hypothesis'. *Plant and Soil,* **342,** 49-57

Palmer, S. C. F. & Truscott, A.-M. (2003a). Browsing by deer on naturally regenerating Scots pine (*Pinus sylvestris* L.) and its effects on sapling growth. *Forest Ecology and Management*, **182**, 31-47.

Palmer, S. C. F. & Truscott, A. M. (2003b) Seasonal habitat use and browsing by deer in Caledonian pinewoods. *Forest Ecology and Management,* **174**, 149-166.

Palmer, S. C. F., R. J. Mitchell, A. M. Truscott & Welch, D. (2004). Regeneration failure in Atlantic oakwoods: the roles of ungulate grazing and invertebrates. *Forest Ecology and Management,* **192,** 251-265.

Pankratov, T. A., Ivanova, A. O., Dedysh, S. N. & Liesack, W. (2011) Bacterial populations and environmental factors controlling cellulose degradation in an acidic *Sphagnum* peat. *Environmental microbiology*, **13**, 1800-14.

Paterson G. (1994) Creating New Native Woodlands. *Forestry Commission Bulletin 112*. HMSO, London.

Pei, M. & Hunter, T. (2000) Integrated control of willow rust in renewable energy plantations in the UK. *Pesticide Outlook*, **11**, 145-148

Penuelas, J., Ogaya, R., Boada, M. & Jump, A.S. (2007) Migration, invasion and decline: changes in recruitment and forest structure in a warming-linked shift of European beech forest in Catalonia (NE Spain). *E*, **30**, 829-837.

Peterken, G.F. & Mountford, E.P. (1996) Effects of drought on beech in Lady Park Wood, an unmanaged mixed deciduous woodland. *Forestry*, **69**, 125-136

Powlson, D.S., Gregory, P.J., Whalley, W.R., Quinton, J.N., Hopkins, D.W., Whitmore, A.P. *et al.* (2011) Soil management in relation to sustainable agriculture and ecosystem services. *Food Policy*, **36**, S72-S87.

Powlson, D.S., Whitmore, A.P. & Goulding, K.W.T. (2010) Soil carbon sequestration for mitigating climate change: distinguishing the genuine from the imaginary. *Handbook of Climate Change and Agroecosystems, ICP Series on Climate Change Impacts, Adaptation Vol. 1. (eds D. Hillel, C. Rosenweig)* pp. 393–402. Imperial College Press, London .

Press, M.C., Woodin, S.J. & J.A. Lee. (1986) The potential importance of an increased atmospheric nitrogen supply to the growth of ombrotrophic *Sphagnum* species. *New Phytologist*, **103**, 45-55.

Pswarayi, A., van Eeuwijk, F.A., Ceccarelli, S., Grando, S., Comadran, J., Russell, J.R., *et al.* (2008) Changes in allele frequencies in landraces, old and modern barley cultivars of marker loci close to QTL for grain yield under high and low input conditions. *Euphytica*, **163**, 435-447

Pyatt, D. G., Ray, D. & Fletcher, J. (2001). *An Ecological Site Classification for Forestry in Great Britain*. Forestry Commission Bulletin 124. Forestry Commission, Edinburgh.

Rodwell, J.S. ed. (1991). *British Plant Communities. Volume 1. Woodlands and scrub.* Cambridge University Press.

Rochette, R. (2008) No-till only increases N_2O emissions in poorly-aerated soils. *Soil & Tillage Research*, **101**, 97–100.

Royal Society of Edinburgh (2011) *Facing up to climate change: breaking the barriers to a low-carbon Scotland. Report of RSE committee of inquiry, March 2011.* Royal Society of Edinburgh, Edinburgh.

Scottish Executive (2006) *The Scottish Forest Strategy*. Forestry Commission Scotland, Edinburgh.

Scottish Executive Environment and Rural Affairs Department (2007) ECOSSE – Estimating Carbon in Organic Soils Sequestration and Emissions. Scottish Executive, Edinburgh

Seufert V., Ramankutty, N. & Foley, J.A. (2012) Comparing the yields of organic and conventional agriculture. *Nature*, **485**, 229-235.

Smart, S., Henrys, P., Scott, W., Hall, J., Evans, C., Crowe, A, *et al.* (2010). Impacts of pollution and climate change on ombrotrophic *Sphagnum* species in the UK: analysis of uncertainties in two empirical niche models. *Climate Research*, **45**, 163-177.

Smith, P., Andrén, O., Karlsson, T., Perälä, P., Regina, K., Rounsevell, M., *et al.* (2005) Carbon sequestration potential in European croplands has been overestimated. *Global Change Biology*, **11**, 2153-2163.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., *et al.* (2007) Agriculture. *Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change* (eds B. Metz, O.R. Davidson, P.R. Bosch, R. Dave, L.A. Meyer) .Cambridge University Press, Cambridge and New York.

Stevenson, A.C., Jones, V.J. & Battarbee, R.W. (1990) The cause of peat erosion : a palaeolimnological approach. *New Phytologist*, **114**, 727-735.

Stunell, J., Jones, A., Wagstaff, S. & Maslen, S. (2010) *Investigating the impacts of windfarm development on peatlands in England: Part 1 Final Report.* Natural England commissioned report from Maslen Environmental, Shipley, UK.

Sun, B., Hallett, P.D., Caul, S., Daniell, T.J. & Hopkins, D.W. (2011) Distribution of soil carbon and microbial biomass in arable soils under different tillage regimes. *Plant & Soil*, **338**, 17-25.

The Scottish Government (2009a) *Scottish Soil Framework*. The Scottish Government, Edinburgh. http://www.scotland.gov.uk/Resource/Doc/273170/0081576.pdf

The Scottish Government (2009b) *The Scottish Government's Rationale for Woodland Expansion*. The Scottish Government, Edinburgh

The Scottish Government (2010a) *A Low Carbon Economic Strategy for Scotland: Scotland – A Low Carbon Society.* The Scottish Government, Edinburgh. http://www.scotland.gov.uk/Publications/2010/11/15085756/0

The Scottish Government (2010b) *A vision for Scottish agriculture*. The Scottish Government, Edinburgh.

The Scottish Government (2010c) *The Scottish Forestry Strategy 2010-2013 Implementation Plan & 2009-2010 Progress Report*. Forestry Commission Scotland, Edinburgh. http://www.scotland.gov.uk/Publications/2010/02/10144335/1

The Scottish Government (2010d) *Scottish Planning Policy*. The Scottish Government, Edinburgh.

The Scottish Government (2011) *Getting the best from our land. A land use strategy for Scotland*. Scottish Government, Edinburgh.

Towers, W., Hester, A.J., Malcolm, A., Hall, J. & Stone, D. (2004). *The potential for native woodland in Scotland: the Native Woodland Model*. Scottish Natural Heritage, Battleby.

Turetsky, M. R., Crow, S. E., Evans, R. J., Vitt, D. H., & Wieder, R. K. (2008) Trade-offs in resource allocation among moss species control decomposition in boreal peatlands. *Journal of Ecology*, **96**, 1297-1305.

Usher, M.B., Sier, A.R.J., Hornung, M. & Millard, P. (2006) Understanding biological diversity in soil: The UK's Soil Biodiversity Research Programme. *Applied Soil Ecology,* **33**, 101–113.

Valentine, J., Clifton-Brown, J., Hastings, A., Robson, P., Allison, G. & Smith, P. (2012) Food vs. fuel: the use of land for lignocellulosic 'next generation' energy crops that minimize competition with primary food production. *Global Change Biology Bioenergy*, **4**, 1-19.

Van Der Heijden, E., Verbeek, S.K. & Kuiper, P.J.C. (2000) Elevated atmospheric CO_2 and increased nitrogen deposition: effects on C and N metabolism and growth of the peat moss *Sphagnum recurvum* P. Beauv. var. *mucronatum* (Russ.) Warnst. *Global Change Biology*, **6**, 201-212.

Van der Heijden, M.G.A., Bardgett, R.D. & van Straalen, N.M. (2008) The unseen majority: soil microbes as drivers of plant diversity and productivity in terrestrial ecosystems. *Ecology Letters,* **11**, 296–310.

Verhoeven, J.T.A. & Liefveld, W.M. (1997) The ecological significance of organochemical compounds in *Sphagnum*. *Acta Botanica Neerlandica*. **46**, 117-130.

Wäckers, F.L. (2004) Assessing the suitability of flowering herbs as parasitoid food sources: flower attractiveness and nectar accessibility. *Biological Control*, **29**, 307-314.

Wallander, H., Ekblad A. & Bergh J. (2011) Growth and carbon sequestration by ectomycorrhizal fungi in intensively fertilized Norway spruce forests. *Forest Ecology and Management*, **262**, 999-1007.

Watts, C.W., Clark, L.J., Poulton, P.R., Powlson, D.S., & Whitmore, A.P. (2006) The role of clay, organic carbon and long-term management on mouldboard plough draught measured on the Broadbalk wheat experiment at Rothamsted. *Soil Use and Management*, **22**, 334–341.

Whitfield S., Reed M., Thomson K., Christie M., Stringer L.C., Quinn C. H., *et al.* (2011) Managing peatland ecosystem services: Current UK policy and future challenges in a changing world. *Scottish Geographical Journal*, **127**, 209-230.

Woodland Expansion Advisory Group (2012) *Report of the Woodland Expansion Advisory Group to the Cabinet Secretary for Rural Affairs and Environment, Richard Lochhead MSP* http://www.forestry.gov.uk/weag

Yeloff, D. E., Labadz, J. C., & Hunt, C. O. (2006) Causes of degradation and erosion of a blanket mire in the southern Pennines, UK. *Mires and Peat*, **1**, 1-18.

Chapter 3: Sustaining Food Production

3.1 Summary

With increasing human populations and associated increases in *per capita* food demands, sustaining food production is a key challenge for policy makers. Crops, livestock, soil formation, and pollination were considered by the Ecosystem Assessment Working Group meeting to be the most important ecosystem services with respect to the broad policy goal of sustaining food production. The UK NEA considers crop and livestock production as provisioning services, soil formation as a supporting service and pollination as a regulating service that may be both a final ecosystem service and an intermediate service/ecosystem process.

With respect to crops:

- Crop production in Scotland is highly mechanised, with considerable inputs (i.e. nutrients, herbicides, pesticides etc.) and a high degree of disturbance and intervention, which means that biodiversity (in particular above-ground biodiversity) is likely to have a limited role in underpinning this ecosystem service.
- The distinction between uplands and lowlands is a biophysical one, and it is the most important determinant for the distribution of arable agriculture in Scotland. Biophysical conditions are a major determinant of crop production.
- Crop production is often seen as in direct conflict with biodiversity conservation, with increased crop productivity associated with decreased biodiversity. However, many studies have found evidence for a positive relationship between diversity and function, where function is usually measured as productivity or as related regulation of pests and diseases by natural enemies. There is a general consensus that this positive relationship is due to the composition of functional traits in the system.
- Studies of arable weeds suggest that a decline in weed diversity is likely to have a detrimental effect on the balance of different functional groups of organisms within the arable food-web. This will have serious consequences for ecosystem functions including decomposition, soil nutrient retention and cycling, pest and disease population control, pollination and, in the long-term, primary productivity and system sustainability.

With respect to livestock:

- Livestock production (i.e. beef, sheep, poultry, pigs, dairying and other livestock products) represents the dominant agricultural sector in Scotland, contributing about 55% of the total value of agricultural production in 2012.
- Livestock production is particularly important for the Scottish uplands, where it is the dominant agricultural land use, utilising a large proportion of Scotland's semi-natural grassland, heath and moor.
- There is a considerable literature on how livestock production affects biodiversity and biophysical processes (both positively and negatively), but there is rather little information on how this provisioning service is supported by biodiversity and biophysical processes, particularly in relation to sustaining food production.
- Despite the limited evidence base, it is clear that biophysical drivers and processes, such as climate, soil type and water flow can impact on livestock production directly by influencing the quality of grazing, and indirectly through regulation of disease and pests.

With respect to pollination:

- Overall, pollination services are estimated to be worth £11.7 billion in the EU (Gallai *et al*. 2009), which equates to c. £615 million per annum to U.K. agriculture, based on U.K.'s 5.26% share of total EU agricultural production (FAOStat, 2009).
- The link between biodiversity and pollination is both strong and clear. Pollination is primarily provided by natural populations of bees and other insects, and therefore a reduction in pollinators can be expected to have a deleterious effect on the provision of this ecosystem service.
- However, a relatively small proportion of current Scottish crop production is dependent on pollination (about 13% of total output value). In addition, since wind-pollinated grasses are the main source of fodder, there is likely to be no impact of pollinator losses on the production of meat and dairy products or on grain production.

With respect to soil formation:

- Soils are vitally important in terms of sustaining food production, with biodiversity and biophysical and biotic processes as key drivers of soil formation and function.
- Soil biota provide many functions which are critical to crop production: soil organisms can confer stability in the face of stress and disturbance, protect against soil-borne diseases, and enhance nutrient use efficiency.
- As for crop production, the links between soil formation, biodiversity, and ecosystem service delivery are complex. There is evidence that soil biodiversity in many areas is in decline, and there is a strong correlation between increasing intensity of land-use and a decline in soil biodiversity. If soil biota are considered a component of a healthy soil, then the impacts of farming practice on soil biota could be considered a negative impact on soil formation.
- Research on the role of soil biodiversity in ecosystem function has lagged behind corresponding research on above-ground organisms, and knowledge of soil biodiversity and its contribution to ecosystem function and the provision of ecosystem services are limited. Functional trait based approaches may be useful in addressing the research gaps in this field.

Overall:

- Although there is evidence of the importance of biodiversity in the delivery of ecosystem services with respect to sustaining food production, it appears that for some services functional diversity (of plants, animals and micro-organisms) is more important than biodiversity itself.
- Natural processes and biodiversity at a range of levels can help to deliver services directly relevant to sustaining food production, and can do so in a sustainable manner. However, it is clear that we need a better understanding of how to integrate nature conservation with food production, and to balance the possible negative as well as positive effects of biotic processes and biodiversity on system delivery in a Scottish context.

3.2 Definition of sustaining food production

Sustaining food production, and ensuring the continuity of food supply to the Scottish people, are key policy goals of The Scottish Government (2009a). World population growth and the increased demand for food, together with other global issues such as climate change, reduced resource availability (e.g. oil), high commodity prices, high price volatility, and changes in consumption patterns with increasing wealth (i.e. more demand for dairy, fish and meat) are all having a major impact on food security (Government Office for Science 2011). Self-sufficiency in food production is not a realistic goal for Scotland. No national figures are available on self-sufficiency, but Scotland consumes much more food than it produces (Scottish Food Forum 2010). However, improving food security by maintaining the capacity for food production, creating resilient food supply chains, and developing more sustainable production systems are important policy goals that are achievable and have been recognised by the Scottish Government (The Scottish Government 2008; 2009a; 2009b; 2009c). This chapter focuses on the broad policy goal of sustaining food production and considers the linkages between biodiversity/biotic and biophysical processes and some of the ecosystem services and 'goods' that are key to achieving this policy goal.

Before moving on, it is important to note that sustaining food production is not the same as *sustainable* food production. The latter explicitly considers the impact of food production on the natural environment, with sustainable production being the production of food with minimal environmental impact. In contrast, sustaining food production, the focus of this chapter, looks at all kinds of production systems (including sustainable farming) as part of the process of delivering sufficient food in the long-term. However, we do not extend this analysis to cover the role of aquaculture and fisheries in sustaining food production.

3.3 Prioritised Ecosystem Services

Unsurprisingly, the two ecosystem services ranked highest during the Ecosystem Approach Working Group workshop with respect to the delivery of the broad policy goal of sustaining food production were crop and livestock production. The other prioritised ecosystem services were pollination and soil formation.

3.3.1 Crops

The provisioning service of crops produces a wide range of 'goods', including food, fibre, and increasingly biomass fuels. Crop production can in turn influence the delivery of many other ecosystem 'goods' and services. For example, crop production can lead to declines in biodiversity and inhibit attempts to halt biodiversity loss. Declines in biodiversity may in turn impact on soil formation and soil C sequestration. In addition, the use of land for food crops can be in conflict with the use of land for forestry, and hence timber-based C sequestration.

In this chapter we focus on those crops that are of direct relevance to food production, being either foodstuffs themselves, or fodder crops for livestock. In this context the provisioning service of crops delivers a wide range of 'goods' including cereal grains, vegetables and fruits. In terms of total agricultural gross output value in 2011, crops and horticulture had a 35% share (cereals 16%, potatoes 8%, oilseed rape 2%) (The Scottish Government 2012c). In Scotland in 2012, there were 457,709 ha of cereals, 41,082 ha of oilseed rape, peas and beans, 29,536 ha of potatoes, 19,823 ha of fodder crops, 15,430 ha of vegetables and 1734 ha of soft fruit (The Scottish Government 2012b). In Scotland, whisky production is an interesting issue. It contributes to Scotland's food and drink agenda and its economy, but it is difficult to argue that the use of cereals for whisky production contributes to sustaining food production.

Crop production in Scotland is essential in order to address issues of food security and sustaining food production in the face of increasing human populations and declining ecosystem functioning.

3.3.2 Livestock

The NEA identifies livestock as a provisioning service (UK NEA Ch. 2), providing 'goods' such as meat, dairy products, eggs, fibre, hide and feathers (UK NEA Ch. 15). Livestock (i.e. beef, sheep, poultry and pigs) represent the dominant agricultural product in Scotland, contributing about 42% of the total value of agricultural production in 2011, with dairying and other livestock products contributing a further 13% (The Scottish Government 2012a).

Despite its importance to the Scottish economy, livestock production in Scotland is in decline. Agricultural census data from the Scottish Government have shown that the national sheep flock declined by over 2.96 million between 1999 and 2012 (from 9.70 million to 6.74 million, a 30.6% reduction), and the number of breeding ewes is now at its lowest level for over 100 years (The Scottish Government 2012b). Mutton and lamb production fell by 11,900 tonnes between 1999 and 2011 (Scottish Executive 2002; The Scottish Government 2012c). Similarly, the beef cattle herd declined by 118,850 between 1999 and 2012 to less than a million animals (The Scottish Government 2012b). The total number of dairy cattle also showed a decline over this period falling by 60,447 (The Scottish Government 2012b) and milk production was 92 million litres lower in 2011 than in 1999 (Scottish Executive 2001; The Scottish Government 2012a). The greatest declines in livestock have been in the hills and uplands of the north and west of Scotland (SAC 2008; Thomson 2011), with important ramifications for the semi-natural habitats of high conservation concern that are maintained by grazing. These declines have been fuelled by a combination of factors, including a general down-turn in the economic viability of hill farms, the foot-and-mouth disease outbreak in 2001, livestock reductions related to agri-environment schemes, changes in the way that livestock farmers are subsidised, and an ageing farming population (SAC 2008). This recent decline in ruminant numbers goes counter to any policy of sustaining food production and highlights how factors such as the Common Agricultural Policy and market forces can affect other policy goals.

3.3.3 Pollination

Pollination is identified in the NEA as both a fundamental ecosystem process/intermediate service and a final ecosystem service. Here, however, we can circumvent debates concerning the precise definition of pollination as an ecosystem service by considering instead the relevant 'good', in this case the pollination of food.

In 2007 20% of UK cropped area was comprised of pollinator-dependent crops (8% in Scotland) and pollination is estimated to be worth £430 million to UK agriculture (£47 million in Scotland). In 2011, the fruit crop produced in Scotland had a value of £94.4 million and the rapeseed oil crop had a value of £53.0 million, which equated to approximately 15.5% of the total output value from crops (The Scottish Government 2012c).

With respect to sustaining food production the 'good' delivered by pollination is mainly confined to lowland systems in Scotland. Food production in upland systems is heavily livestock-based and dependent on wind-pollinated grass species, even in semi-natural habitats. Even within lowland systems pollinators are not important in the production of cereals, or the fodder species such as *Lolium perenne* which are important in intensive livestock systems.

Although a relatively small proportion of the current Scottish crop production is dependent on pollination, and the crops supported by pollinators are not major components of our diet, they are some of the most profitable crops (for example soft fruits such as raspberries, strawberries and blackcurrants) and so are of great economic importance despite being relatively less important for the broad policy goal of sustaining food production.

3.3.4 Soil formation

In the UK NEA, soil formation (the formation and degradation of the UK soil resource) is classified as one of the supporting services (along with primary production, nutrient cycling and water cycling). Soil formation is vitally important for other services such as food production (arable crops and pasture), timber production, and carbon storage and sequestration (particularly in peat and other organic soils). With respect to the broad policy goal of sustaining food production, we are interested here in the role of soil in providing a medium for the growth of either food or fodder crops. This is in contrast to our consideration in Ch. 2 (Low Carbon Economy) of soil formation as the basis for C sequestration.

The NEA states that “soil loss affects agricultural production and nutrient availability (Quinton *et al*. 2010), and also water quality (Environment Agency 2004). This also causes a loss of soil biodiversity and the ecosystem services that it underpins…”.

3.4 How do biodiversity and biotic and biophysical processes underpin these services and ‘goods’?

3.4.1. Crops

Perhaps one of the most obvious roles of biodiversity in promoting crop production is the provision of plant species and genetic material for crop varietal improvement and increased crop productivity. This process has been on-going since humans started farming land for crops in the Neolithic age. However, for many crops this process is now highly regulated, laboratory-based, and dependent on existing crop collections. Consequently it is less dependent on wild diversity, and the direct interaction of any given crop and its environment (i.e. the “natural” acquisition of novel genes by the crop from the wider wild gene pool, as undertaken for example in bioprospecting as discussed in Section 4.4.2). Declines in wider plant biodiversity in, for example, Scottish arable systems, are therefore unlikely to limit the potential for continued productivity improvements through crop breeding.

However, within-crop genetic diversity can still play an important role in promoting crop productivity. In recent decades, crop production has been dramatically increased by large inputs of fertilisers, water and pesticides, which now typify the intensive management regimes of modern agriculture. Along with these developments has been a trend for the growth of crops in genetically homogeneous stands. Although reduced genetic diversity can simplify processes such as harvesting, for example by unifying crop ripening times, increasing within-field crop genetic diversity can improve crop yield through plant complementarity and facilitation (beneficial interactions between neighbouring plants). Avoidance of crop monocultures over large scale landscapes and rejecting over-simplistic crop protection measures will also reduce the promotion of those pest and disease biotypes/pathotypes which are capable of overcoming current crop protection measures (Birch *et al.* 2011). Mixtures of different crop genotypes within an arable field can increase productivity through niche differentiation, resulting in better exploitation of the available resources, and through a reduction in the ability of pests and diseases to spread through the field populations (Newton *et al*. 2009). Productivity can also be enhanced by improved competitiveness with weeds (resulting in turn in greater niche exploitation), and through stabilised yields (Kiær *et al.* 2012).

In terms of wider (i.e. non-crop) diversity, crop production is often assumed to be in direct conflict with nature conservation. The NEA argues that there is little evidence that increased non-crop biodiversity is important for increased crop productivity (UK NEA Ch. 15), rather that yields and food production have increased in spite of declines in farmland biodiversity, and that, in general, many interactions between provisioning services and other ecosystem services (such as nature conservation) are negative. For example, the diversity of soil microbial communities (Daniell *et al.* 2001), non-crop arable vegetation (Gabriel *et al.* 2005; Hawes *et al.* 2010) and lowland grassland vegetation (Klaus *et al.* 2011) is typically low in high intensity agricultural systems.

Although to date the declines in, for example, soil biodiversity do not appear to have had negative consequences for crop production (or at least, crop production has continued to increase despite declines in soil biodiversity), in the longer term they may inhibit our ability to move to more sustainable agricultural production, and thus to sustain food production. The UK NEA states that “care will need to be taken to ensure that this increased level of management [associated with a drive for increased crop yields] does not have adverse impacts on biodiversity and ecosystem functioning". In addition, and despite the historically negative production-biodiversity relationship, there is evidence that wider species diversity within crop systems can enhance crop productivity, particularly if the use of mechanisation or agro-chemicals is restricted (as discussed in Section 2.3.4). The non-crop components of the system could positively influence crop production through changes in primary productivity (e.g. beneficial microbes and symbionts promoting crop growth), increased pollination efficiency, improved biocontrol of pests and pathogens, and more effective nutrient cycling, but such benefits may be greatest when switching to a more sustainable production system.

Such relationships between biodiversity and crop production can be complex, and mediated through chains and webs of biotic interactions. For example, populations of insects responsible for pest biocontrol and pollination in agricultural systems might be less effective in crop monocultures due to reduced availability and quality of plant food resources (Biesmeijer *et al.* 2006; Wackers *et al.* 2008). This effect arises from the combined effects of habitat loss, habitat fragmentation and intensive agricultural practices (Kremen *et al.* 2002) on vegetation diversity and abundance. Diversity in the weed layer results in increased diversity of herbivore consumers (Marshall *et al*. 2003), an effect then propagated through the food web to the third (predator) trophic level (e.g. Hawes *et al*. 2003, 2009; Taylor *et al*. 2006). Weeds, and in particular dicotyledonous weed species, represent a highly valuable resource to primary consumers within and around arable fields, supporting up to ten times the biomass of herbivores per unit plant mass compared to crop plants (Hawes, unpublished data). Consequently arable weeds are increasingly seen as an important driver of biodiversity at a number of levels, crucial to the functioning of arable systems, providing a greater variety of form, composition and function in higher trophic levels than the few crop species that dominate arable land (Hawes *et al*. 2003; Norris & Kogan 2000). Declining diversity of the weeds on which these food-webs are based is therefore likely to have a detrimental effect on the balance of different functional types of organisms within the arable food-web, with serious consequences for ecosystem functions including decomposition, soil nutrient retention and cycling, pest and disease population control, and ultimately, in the long-term, primary productivity and system sustainability (Loreau 1998; Hector *et al*. 2001; Scherer-Lorenzen *et al*. 2003).

The NEA acknowledges that several authors argue in favour of the importance of biodiversity for ecosystem productivity (UK NEA Ch. 15). Declining biodiversity in managed systems has raised concerns about the potential for loss of system function (Marshall *et al*. 2003). Many published studies have found evidence for a positive relationship between diversity and function, where function is usually measured as productivity (Diaz *et al*. 2003), and these relationships and their underlying mechanisms are clearly relevant to crop production. The mechanisms responsible for these relationships are still the subject of debate, although there is a general consensus that this positive relationship is due to the diversity of functional traits represented by organisms in the system rather than the biodiversity of the system *per se* (Bolnick *et al*. 2003).

Overall, the balance between wider biodiversity within the landscape and the delivery of crop production appears delicate and complex. The NEA presents both sides of the argument: (a) biodiversity and many biotic processes are not necessary for production (witness the negative correlation over the past 50 years between production and biodiversity, e.g. Pimm *et al*. 1995), countered by (b) biodiversity and many biotic processes are necessary for systems to function sustainably in the long-term (Loreau *et al*. 2001 and many others). Hence, biodiversity and biotic processes become more important for sustaining food production *in toto* as production systems get more sustainable and/or less intensive (as discussed in Section 2.4.4).

In contrast to the uncertain role biodiversity and biotic processes play in sustaining crop production, biophysical processes play a fundamental role and levels of productivity are driven, at least in part, by soil and hydrological processes and weather conditions. There are many physical and bio-chemical processes involved in soil formation and functioning which are essential if crop production levels are to be sustained, including processes such as nitrification and mineralisation. Chemical processes in the soil such as acidification, leaching, nutrient depletion, salinisation and alkalisation can lead to reductions in crop production. Physical processes such as crusting and compaction of the soil can lead to poor root penetration, low infiltration of water, reduced aeration and anaerobiosis, all of which can lead to reduced productivity. Soil erosion by both surface water runoff and wind can lead to soil degradation and reduced crop productivity.

3.4.2 Livestock

As with crop production, the most obvious direct impact of biophysical processes on livestock production is the limitation set by biophysical conditions on primary production. Indeed, the focus of agriculture in many areas on livestock and grazing is itself a consequence of biophysical limitations on system productivity: if the land were more productive there is a high probability that it would instead be used for crop production (as evidenced by the switch from crop production to grazing following historic cooling of climate; Brooker 2011).

In upland systems relatively low productivity necessitates extensive grazing and dependency on semi-natural habitats (which in turn creates a feedback loop whereby the persistence of these habitats becomes dependent on grazing). Scotland is home to the majority of the UK's mountains, moorlands and heaths, which cover 44% of Scotland's land area. This land, like semi-natural grassland which cover 19% of Scotland, is wetter and colder and has lower agricultural potential than enclosed farmland, which covers 19% of Scotland (compared to 41% of England). Mountains, moorlands, heaths and semi-natural grasslands all support grazing (UK NEA Ch. 2), and much of the production of ruminant livestock is pasture based (both semi-natural and improved), particularly in the uplands. This explains why livestock represent the dominant agricultural product in Scotland. In lowland systems, in contrast, higher productivity enables more intensive grazing systems based around the production of high-input fodder crops, and this is economically sustainable because of the greater cash value of the derived 'goods' (i.e. beef and dairy products).

Although the impacts of livestock production on biodiversity have been well studied, there has perhaps been less consideration of the way in which biodiversity and biotic processes might support livestock production. However, the contrasting livestock production systems found in the Scottish uplands and lowlands enables us to make some broad predictions concerning this relationship. In upland systems there is clearly a greater direct dependency of livestock production on semi-natural habitats, although whether productivity levels and stability of production in these habitats is – on average – likely to be promoted by increased biodiversity or status of these systems is unclear. Habitat diversity may have beneficial impacts on extensive livestock production. Habitat diversity will likely lead to increased fodder diversity, which has been shown to increase livestock production (Wang *et al.* 2010) with different habitats providing a range of fodder which is either optimal or available at different times of year, thus helping to sustain grazing year-round. Habitat diversity may also provide areas of shelter (e.g. silvopastoral systems and wood pasture) which could improve animal welfare and potentially increase productivity; benefits of habitat diversity have been shown for wild ungulates such as North American elk *Cervus Canadensis* (Sawyer *et al.* 2007). However, habitat diversity also has disadvantages. For example, habitat diversity may increase the risk to livestock of ingesting or coming into contact with toxic plants such as Bog Asphodel (*Narthecium ossifragum*) which causes photosensitization.

In contrast, in intensive lowland livestock production systems - and in parallel with intensive crop systems - the roles of biodiversity and biotic processes have been reduced by the capacity to use artificial inputs and mechanisation and/or temporal separation of fodder production from consumption (e.g. through silage production), and by the direct negative impacts of these management actions on biodiversity (UK NEA Ch. 7). Indeed, and again as for crop systems, management in these lowland areas is in some cases targeted towards reducing biodiversity, for example through the production of single-species fodder crops such as *Lolium perenne* via high nutrient inputs and re-seeding.

One aspect of the regulation of livestock production by biota that has been well studied is the occurrence of pests and diseases. However, this biotic process is again strongly influenced by both the biotic and abiotic environment.

Climate affects the range of infectious diseases that occur, whereas weather affects the timing and intensity of disease outbreaks (Bezirtzoglou *et al*. 2011). A climate-driven model explains, in both space and time, many aspects of the vector borne disease Bluetongue, including the 2006 outbreak in northwest Europe, which affected sheep and cattle and led to the implementation of a mandatory vaccination campaign in Scotland (Guis *et al*. 2012). The Shetlands were exempt from certain regulations under The Bluetongue (Scotland) Order 2008, because of their biophysical conditions, i.e. location, wind and temperature, which meant that midges were not an issue on the islands, which highlights the major role of biophysical underpinning on diseases and pests. Recently, a new disease, caused by the Schmallenberg virus, emerged in North-western Europe and the UK, with devastating impact on unborn lambs. Like Bluetongue virus, this new virus is transmitted by midges (Cullicoids), which depend on the presence of suitable habitat, temperature and moisture (Gibbens 2012). Climate change also has an impact on the epidemiology of helminth parasites and trematodes, with negative implications for sheep health and welfare and the efficiency of food production in Scotland (Kenyon *et al*. 2009). The incidence of liver fluke, a pathogen of livestock, wildlife and humans, has been increasing in the UK with unprecedented levels of future liver fluke disease risk predicted in parts of the UK (Fox *et al*. 2011; McCann *et al*. 2010). The risk of liver fluke is also affected by rainfall, temperature, soil type (low pH), slope, and wet grazing conditions (McCann *et al*. 2010; Pritchard *et al*. 2005). Management prescriptions associated with agri-environment schemes such as the Environmentally Sensitive Areas Scheme have encouraged wetter grassland conditions for breeding and migrating birds and invertebrates; these actions also maintain suitable habitats for endangered flora. The maintenance of water flow in drainage ditches and dykes, and the associated accumulation of surface water and ‘ponding’ on adjacent grazing pastures, inevitably favours the reproduction and survival of the intermediate host snail *L. truncatula* and the parasites it carries and propagates, e.g. the liver fluke *F. hepatica* (Pritchard *et al*. 2005). Periods of low rainfall followed by periods of high rainfall increase the risk of waterborne disease outbreaks caused by bacteria and protozoa that are carried by livestock and that are pathogenic to humans, e.g. *E. coli* or *Cryptosporidium* (Nichols *et al*. 2009).

High environmental diversity may increase the risk of diseases/pests in livestock by increasing the number of wild vectors and intermediate hosts (e.g. ticks, midges, snails) and wildlife hosts that share pathogens with livestock. In upland systems increased habitat diversity can lead to increased populations of wild mammals such as rabbits, hares and deer which carry bacteria, viruses and parasites that also affect livestock, for example Johne’s disease, liver fluke, and tick borne diseases (Böhm *et al.* 2007; Smith *et al.* 2009; UK NEA Ch. 15). Greater habitat diversity may also lead to increased levels of parasites, such as liver fluke, by providing habitat for the intermediate host snails (Pritchard *et al.* 2005). Scientific research, as well as experience from veterinarians and farmers, suggests that an homogenous landscape is likely to be easier to manage in terms of pest and sdisease regulation. For example, there is evidence that outdoor reared pigs pose a higher risk of *Toxoplasma* infection than indoor reared pigs, because the natural environment may be contaminated with parasites that are not found in an indoor environment (Van der Giessen *et al.* 2007). As with the production of fodder, it is also possible that pest and disease regulation may be less heavily dependent on natural processes in more intensively managed systems as again, technology and intervention (e.g. all-in-all-out systems, inoculation etc.) in more intensive livestock systems might intervene in the relationship (not least because of the greater relative cash value of the ‘goods’).

A final aspect of biodiversity/biotic processes that is critical to sustaining livestock production is the within-species genetic diversity that exists within livestock (Simm *et al*. 2004). For example, we have sheep and cattle breeds that are highly productive in lowland systems, but that would not survive in upland conditions and similarly, we have upland breeds such as the Scottish Blackface sheep which are not sufficiently productive to justify their use on lowland systems, but that are highly adapted to the hill environment. In some situations livestock have become highly adapted to their environment, for example on North Ronaldsay the sheep have adapted to eat seaweed. Scientists have been studying specific sheep populations, for example the St. Kilda sheep, to get a handle on natural resilience and resistance to diseases. These sheep have to survive without human intervention, so there is more selection pressure on health traits than in intensively managed production systems. Even in lowland systems, resilience to parasite infestation is being studied. Historically, breeds of sheep and cattle have been selected for productivity under the prevailing environmental conditions which includes varying degrees of tolerance to endemic pathogens. Breeds currently prevalent in Scotland may not be best suited to the environment of the future. A rapidly changing environment (warmer, wetter) and pathogen threat will require changes to the genetic makeup of flocks and herds through selective breeding, cross-breeding or the introduction of new breeds.

Overall, and as for crop production, if sustained food production in the long term necessitates a move toward sustainable food production, then the dependency of livestock production on biodiversity, natural systems and biotic processes will necessarily increase.

Pollination

We have already noted that the role of pollination in sustaining food production is likely to be heavily biased toward the production of a limited number of high-value crops within productive lowland systems, for example soft fruits (e.g. raspberry), orchard fruit (e.g. apples) and oilseed rape. However, within this context the role of pollination, and the dependency of food production on underlying biodiversity and biophysical processes, is both obvious and strong.

Biophysical processes, particularly those related to weather and climate, can have a major impact on pollinator numbers and hence on crop productivity. As pollination is primarily provided by natural populations of bees and other insects (Garibaldi *et al.* 2013), a reduction in natural pollinator diversity and numbers can be expected to have a deleterious effect on the provision of this ecosystem service. Declining insect pollinator numbers in the UK and worldwide (Goulson *et al*. 2008) have the potential to seriously affect the productivity of insect-pollinated crops. The annual global economic value of insect pollination is estimated to be $153 billion (Gallai *et al*. 2009) and loss of pollinator services could reduce worldwide crop production up to 8%, necessitating greater agricultural intensification (Aizen *et al*. 2009).

Broader invertebrate diversity is important in providing pollinators (Garibaldi *et al.* 2013), and this in turn is supported by habitat diversity. The UK NEA states that “The majority of pollination services are provided by wild bees and other insects. Diversity within wild communities of pollinators provides resilience against environmental change and can be supported through the provision of natural and semi-natural habitats, including agri-environment schemes, which provide a range of flower communities.” Pollinators are often resource-limited and major declines are driven by both land-use and climate change (Williams *et al*. 2009). A major cause of pollinator decline is likely to be the reduction in the availability and quality of plant food resources (Biesmeijer *et al*. 2006), through the combined effects of habitat loss, habitat fragmentation and intensive agricultural practices (Kremen *et al*. 2002) on vegetation diversity and abundance. In Scotland’s lowland arable-grass system low intensity management systems that increase within-field and field margin resource availability and quality for pollinators, including extended seasonal resource abundance, are likely to promote pollinator diversity through niche differentiation. Additionally, variation in habitat quality at the landscape scale, for example through the introduction of semi-natural habitats and more diverse cropping in the arable system, may enhance pollinator diversity as a consequence of species coexistence mechanisms. However, there is little quantitative data on the impact of alternative land management strategies, even within the arable system, to inform policy and to advise landowners and conservation groups on improving insect pollinator diversity, abundance and pollination services.

Despite the obvious dependency of pollination on natural insect populations, farming practices have continued to have negative impacts on wild insect pollinators, including pesticides, lack of floral food resources, risk of bee pathogens during mass rearing, transporting bees over long distances. Many species of the main insect pollinator groups are listed as priority species in the UK. For example, the majority of bumblebees, which are important pollinators of commercial crops (e.g. soft fruit, oilseed rape) and wild plants, have shown recent population declines, resulting in farmers' buying in pollinators (mainly bumblebees) from commercial sources at considerable expense. The UK hosts 24 species of bumblebees, three other species are now extinct. Seven species of bumblebees are listed as UK priority species and two of them, the great yellow bumblebee (*Bombus distinguendus*) and the shrill carder bee (*Bombus sylvarum*), are particularly under threat. Scotland remains, nonetheless, a stronghold for many bumblebee species (Goulson 2007).

We have suggested that for the services of crop and livestock production, particularly in lowland areas, it may become increasingly important to farm sustainably in order to maintain these services and hence sustain food production. However, in the case of pollination, and the foodstuffs that this service supports, there is clearly already a need for sustainable farming practice in those farming regions where insect pollination is vital in order to support the continued delivery of this ecosystem service and its associated 'goods'.

Soil formation

Chapter 13 of the UK NEA states that "soil formation is a continuous process and its speed and nature is affected by several factors (Jenny 1941) including the parent material, climate, topography, biota (including plants, animals and microorganisms) and land management". With respect to sustaining food production, soil formation's supporting service role results in the provision of a suitable and sustainable plant growing medium. This includes the provision and maintenance of a matrix (through the weathering of rock and accumulation of organic matter) within which plants can grow, the turnover and release of nutrients such as N and P, and the retention and hence provision of water.

As already outlined with respect to C sequestration (Ch. 2), biotic and biophysical processes are essential components and drivers of soil formation and function, and soil organisms are clearly central to these processes. Soil organisms are extremely diverse, and they contribute to a wide range of ecosystem services that are essential to the sustainable functioning of ecosystems (Barrios 2007). Overall reduction in the diversity of some soil organismal groups has been predicted to lead to the loss of ecosystem functions (He *et al*. 2009), but this may be dependent on the specific function and the diversity of soil organisms that underpin it. Some of the soil processes that benefit crop production are carried out by a wide range of organisms, and so there is thought to be a high degree of functional redundancy in the soil community with respect to these "broad" processes (Powlson *et al*. 2011). They include decomposition, transformation of nutrients, mineralisation and the stabilisation of soil structure. These processes can be maintained, even under harsh environmental conditions, because the functions are distributed amongst a large range of soil organisms. In contrast ''narrow'' processes (Powlson *et al*. 2011), where functions are restricted to limited groups of soil organisms or particular environmental conditions, are much less resilient and are more easily reduced or lost (Bardgett *et al*. 2005). For example, the degradation of soils can lead to the loss of predators and biotic regulation, and an increase in pathogens and pests (Sylvain & Wall 2011).

Some soil processes driven by soil organisms, such as nitrification by bacteria, may not be immediately beneficial to crop production. However, negative short-term effects may be traded off against gains for long-term system sustainability. Verbruggen *et al*. (2012) examined the provision of ecosystem services by soil communities from agricultural fields with different management regimes. They found that the soil communities from the different fields varied in their impact on plant productivity and nutrient leaching losses, and but also that there is a potential trade-off between the positive effects of soil communities on sustainability and their negative effects on crop productivity.

The organic matter content of the soil is critical in determining soil properties and function. The NEA (Ch. 13) states that "management strategies aimed at maintaining, or enhancing, the accumulation of soil organic matter can have multiple synergies for provisioning, supporting and regulating services (Lal 2008; Smith *et al*. 2008; Woodward *et al*. 2009). For example, an increase in soil organic matter content can afford benefits for soil fertility such as improved soil structure and water-holding capacity, greater complexity and diversity of the soil food web, binding and transforming pollutants that might otherwise enter the food chain or water supplies, and increased storage and retention of nutrients and water (Lal 2008; Woodward *et al*. 2009)".

As noted in Ch.2 (with respect to peat formation) there is a balance between biotic and abiotic soil processes which regulates the rate of decomposition of organic matter and its subsequent accumulation in soil. Changes in soil organic matter content can be driven by above-ground as well as below-ground biotic processes and biodiversity. For example, there are benefits of planting high-diversity grassland mixtures over monocultures on degraded soils. The NEA states that "increased soil organic matter accumulation can have synergies with biodiversity conservation: planting of high-diversity mixtures of native grassland perennials on degraded, low organic matter content soils can yield advantages over monocultures in terms of productivity, reduced greenhouse gas emissions and carbon storage (Tilman *et al*. 2006), with additional benefits for wildlife conservation".

However, such benefits may be driven by a sampling effect (the increased probability of a more productive species or functional type being present with increasing species number) rather than diversity *per se*, and it is critical to understand which elements of biodiversity – both above- and below-ground - actually contribute to changes in productivity or function. Brussaard *et al*. (2007) found evidence for a number of roles of soil biodiversity which could have positive effects on crop or fodder production. These include conferring protection against soil-borne diseases, system resistance to stress and disturbance, and enhanced nutrient use efficiency (particularly in relation to enhanced mycorrhizal diversity). However, the mechanisms underlying these relationships were not fully understood. Bowker *et al*. (2010) state that research on the role of soil biodiversity and biotic processes in ecosystem function has lagged behind corresponding research on aboveground organisms, and knowledge of soil biodiversity and its contribution to ecosystem function and the provision of ecosystem services is limited. Approaches based on plant (animal and microbe) functional diversity, may be key to understanding soil formation processes (Díaz *et al*. 2007; De Deyn *et al*. 2009), and trait-based assessments of diversity may be more informative than taxonomically based ones in terms of understanding the role of diversity in underpinning ''broad'' and "narrow" functions (Powlson *et al*. 2011). The problem of understanding biodiversity-function relationships may be exacerbated with respect to soil organisms because of the difficulties in identifying species, a lack of knowledge about what they do in soil ecosystems, problems with estimating their biodiversity, and the inability to culture the majority of the organisms for experiments (Bowker *et al*. 2010).

Irrespective of the substantial knowledge gaps concerning the links between biodiversity and biotic processes and soil processes and formation, it is clear that management practice can have substantial effects on soil biota which then influence service provision. Soil biodiversity in many areas is in decline, and there is evidence of a strong correlation between increasing intensity of land-use and this decline in soil biodiversity, which in turn will lead to a reduction in the ecosystem services provided by the soil biota; consequently the extensive exploitation of soils by man can be considered the greatest threat to soil biodiversity (Jeffery & Gardi 2010). But as for crop production, depending on our capacity and willingness (related to economic viability, which is probably greater in the Scottish lowlands), some natural processes such as disease and pest control can be replaced by anthropogenic ones, for example the use of agrochemicals (Sylvain & Wall 2011). But with more sustainable management systems it may not be necessary to replace lost functions and services: for example, the appropriate use of organic fertilisers can enhance the accumulation of soil organic matter, and the use of different crop rotations with ley crops and cover crops can increase soil organic matter and reduce erosion. Minimal tilling and seed-bed preparation, as well as appropriate grazing management, can also reduce soil erosion.

Overall, the link between soil formation and biodiversity and biotic processes is complex. Unravelling this complexity will include understanding which elements of the biota contribute to function, or whether it is their absolute diversity that is critical, and how these relationships are moderated by changes in the abiotic environment (UK NEA Ch. 13). But despite this complexity it seems that declines in diversity and abundance of both above and below-ground organisms can have negative impacts for the various processes that are involved in soil formation. Until this complexity is unpicked, conservation of biodiversity in production systems seems sensible, but can be promoted by more sustainable management practice.

3.5 Interactions between Ecosystem Services

As for services relevant to a low carbon economy (Ch. 2), there are obvious potential conflicts between the services of crop and livestock production in terms of the possible alternative uses of any single parcel of land. Conflicts also exist between food production and other land uses, for example forestry. However, we suggest that the interactions and potential conflicts are not as intense or difficult to resolve *within* the broad policy goal of sustaining food production because underlying biophysical conditions set limits on the potential use of land for certain services. Specifically, in the uplands there is no conflict between livestock and crop production, as the soils and climate are less suitable for crops. In the lowlands, however, there may be increasing conflict if sustaining food production is delivered through systems based on the principle of sustainable food production. Food production per unit area via intensive livestock and associated fodder production systems is much less sustainable than through, for example, cereal production for direct human consumption. Consequently enhanced sustainability may in the future lead to increasingly negative interactions between the services of crop and livestock production in the lowlands.

However, more immediate interactions are clearly occurring between the provisioning services of crop and livestock production, and the supporting services of pollination and soil formation. In lowland systems some of these interactions between services may be complex and context dependent. For example, in some cases honeybees (pollinators) can interact positively with natural enemies (predators and parasitoids) to reduce local pest populations when visiting companion plants as a food resource (Carreck & Williams 2002; Tautz & Rostas 2008), although competition for floral food resources by these two functional groups has also been shown (Wackers & van Rijn 2012). Overall in the lowlands, however, interactions between services tend to be negative: high underlying system productivity (good soils and climate) enables intensive food production, which in turn negatively impacts on soil and pollinator diversity, inhibiting soil formation and pollination. This in turn creates a negative feedback loop to crop and livestock production, and a system imbalance which must be overcome by investment in mechanisation and intensification. Furthermore, agricultural soils are significant sources of anthropogenic pollutants (e.g. Sutton *et al.* 2011), leading to indirect negative effects on the 'goods' and services arising from cropped systems.

The landscape and biodiversity of much of upland Scotland have been shaped by livestock farming for many centuries (Brooker 2011). It has influenced the patterns of settlement, and has largely determined the proportion of open to afforested land. Although heavy grazing has contributed to the widespread decline of woodland and woody shrubs (such as heather moorland) over many hundreds of years, the converse can also happen: declines in grazing in recent years have had negative impacts on the conservation of some open habitats and species of conservation concern, such as species-rich grasslands (Renwick & Waterhouse 2008). Grazing animals are an important tool in the management of non-wooded upland systems for biodiversity conservation and landscape character, and therefore in some parts of Scotland conservation and natural heritage objectives depend on the continuation of livestock farming to some degree. Controlled grazing is involved in the management of many designated sites (SSSIs and Natura 2000 sites). Appropriate management of grazing livestock can benefit wildlife, and without livestock farming, the appearance of the landscape would change, potentially becoming less diverse and for many people less attractive (Morgan-Davies *et al*. 2008). Livestock farming is also an integral part of the culture and history of rural Scotland, and is currently considered vital to the rural economy. The social cohesion of rural communities in the uplands of Scotland is also tightly bound to livestock farming (Holland *et al*. 2011).

Potentially beneficial interactions between food production and biodiversity conservation might be achievable in lowland systems if there is a switch toward more sustainable food production systems. Indeed part of the point of such systems, along with reducing the need for costly mechanisation, is to deliver food with reduced negative environmental impacts (although the two drivers clearly are not mutually exclusive). Without such a switch, intensive farming practices will continue to impact on the delivery of multiple ecosystem services in the wider countryside. For example, the loss of pollinators has implications for the conservation and functioning of non-cultivated habitats (Kevan & Phillips 2001) and for population sizes of wild plant species. In Britain, parallel declines in pollinators and insect-pollinated plants have been demonstrated (Biesmeijer *et al*. 2006) and a reduction in pollination and seed set of fragmented native plant populations has been reported (Rathcke & Jules 1993). Arable vascular plants are particularly vulnerable: between the mid 20th Century and the period from 1987-99, the distribution of a quarter of the vascular plants in Scotland decreased, predominantly arable plants and species associated with grassland and upland habitats (Preston *et al*. 2006). Rare plants in arable or natural habitats may share pollinators with more common plant species and therefore may depend on the management of the common plants in their community for provision of insect pollinators (Gibson *et al*. 2006).

Finally, we can see the substantial impacts of external socio-economic drivers on service delivery, and the associated network of ecosystem service interactions. The recent decline in ruminant numbers goes counter to any policy of sustaining food production and highlights how factors such as the Common Agricultural Policy and market forces can massively affect other policy goals.

3.6 Knowledge gaps

With respect to crop production, we can see that crop production can be sustained with reducing biodiversity, but that enhanced biodiversity can potentially enhance and stabilise crop production in certain circumstances. What is critical is understanding whether sustainable farming systems which enhance and benefit from biodiversity can deliver the sustained food production that currently we believe is only possible through intensive (low biodiversity) land management. This balance is central to making decisions concerning the apparent trade-off between food production and biodiversity conservation. Alongside this comparison of sustainable farming systems which enhance on-farm biodiversity at the expense of yield and intensive production systems with low biodiversity, the relative advantages and disadvantages of land sparing (i.e. the strategy of increasing yields at the cost of biodiversity in some areas to allow land to be spared for conservation elsewhere) versus land sharing also need to be examined in more depth, and at a landscape scale.

With respect to livestock production, overall there is rather little information on how this provisioning service is supported by biodiversity and biophysical processes, particularly in relation to sustaining food production. In addition it is clear that there are potentially complex relationships between the composition of the landscape mosaic and the impacts on livestock production. For example, habitat diversity can provide shelter and multi forage sources, but may also promote pest and disease organisms and their vectors. Understanding this balance is important for integrating potentially conflicting services such as pest and disease regulation and nature conservation. In addition there are substantial opportunities in better understanding the genetic potential within native breeds as a source of material for improved livestock breeding.

With respect to pollination there is – with one or two exceptions (e.g. wildflower strips) - little quantitative data on the impact of alternative land management strategies, even within the arable system, to inform policy and to advise landowners and conservation groups on improving insect pollinator diversity, abundance and pollination services. Ultimately we need to explore, particularly for production systems with a high demand for pollination, the design of mosaic environments that promote pollinator diversity whilst limiting pests and diseases.

With respect to soil formation many studies indicate that there is insufficient information to understand properly the importance of biodiversity (e.g. soil organism diversity) for this regulating service, and to stabilise associated ecosystem service provision (Bardgett & Wardle 2010; Loreau 2010). Despite considerable recent concerted efforts in this field (e.g. Usher *et al.* 2006) there are many remaining knowledge gaps, not least because of the complexity of soil systems and function, the high level of soil organism diversity, and the context-dependency of many soil processes. Key targets would appear to be separating out “broad” from “narrow” functions, and understanding whether it is particular functional groups or diversity *per se* that is critical for their regulation.

References

Aizen, M.A., Garibaldi, L.A., Cunningham, S.A. & Klein, A.M. (2009) How much does agriculture depend on pollinators? Lessons from long-term trends in crop production. *Annals of Botany*, **103**, 1579-1588.

Bardgett, R.D. & Wardle, D.A. (2010) Aboveground-belowground linkages: Biotic interactions, ecosystem processes, and global change. *Oxford University Press, Oxford.*

Bardgett, R.D., Usher, M.B. & Hopkins, D.W. (2005) *Biodiversity and Function in Soils*. British Ecological Society Ecological Reviews, Cambridge University Press, Cambridge.

Barrios, E. (2007) Soil biota, ecosystem services and land productivity. *Ecological Economics*, **64**, 269-285.

Bezirtzoglou, C., Dekas, K. & Charavlos, E. (2011) Climate changes, environment and infection: Facts, scenarios and growing awareness from the public health community within Europe. *Anaerobe*, **17**, 337-340.

Biesmeijer, J.C., Roberts, S.P.M., Reemer, M., Ohlemüller, R., Edwards M., Peeters, T., *et al.* (2006) Parallel declines in pollinators and insect-pollinated plants in Britain and the Netherlands. *Science*, **313**, 351-354.

Birch, A.N.E., Begg, G.S. & Squire, G.R. (2011) How agro-ecological research helps to address food security issues under new IPM and pesticide reduction policies for global crop production systems. *Journal of Experimental Botany*, **62**, 3251-3261.

Bolnick, D.I., Svanback, R., Fordyce, J.A., Yang, L.H., Davis, J.M., Hulsey, C.D., *et al.* (2003) The ecology of individuals: incidence and implications of individual specialisation. *The American Naturalist*, **161**, 1-28.

Bowker, M.A., Maestre, F.T. & Escolar, C. (2010) Biological crusts as a model for examining the biodiversity-ecosystem function relationship in soils. *Soil Biology and Biochemistry*, **42**, 405-417.

Brooker, R.W. (2011) The changing nature of Scotland's uplands - an interplay of processes and timescales. *The Changing Nature of Scotland* (eds. Marrs, S., Foster, S., Hendrie, C., Mackey, E.C., Thompson, D.B.A.) pp 381-396. Scottish Natural Heritage, Edinburgh.

Brussaard, L. Pulleman, M.M., Ouédraoga, E., Mando, A. & Six, J. (2007) Soil fauna and soil function in the fabric of the food web. *Pedobiologia*, **50**, 447-462.

Carreck, N.L. & Williams, I.H. (2002) Food for insect pollinators on farmland: Insect visits to the flowers of annual seed mixtures. *Journal of Insect Conservation*, **6**, 13-23.

Daniell, T.J., Husband, R., Fitter, A.H. & Young, J.P.W. (2001) Molecular diversity of arbuscular mycorrhizal fungi colonising arable crops. *FEMS Microbiology Ecology*, **36**, 203-209.

De Deyn, G.B., Quirk, H., Zho, Y., Oakley, S., Ostle, N.J. & Bardgett, R.D. (2009) Vegetation composition promotes carbon and nitrogen storage in model grassland communities of contrasting soil fertility. *Journal of Ecology*, **97**, 864–875.

Díaz, S., Lavorel, S., de Bello, F., Quétier, F., Grigulis, K. & Robson, T.M. (2007) Incorporating plant functional diversity effects in ecosystem service assessments. *Proceedings of the National Academy of Sciences of the USA*, **104**, 20684–20689.

Díaz, S., Symstad, A.J., Chapin III, F.S., Wardle, D.A. & Huenneke, L.F. (2003) Functional diversity revealed by removal experiments. *Trends in Ecology and Evolution*, **18**, 140-146.

Environment Agency (2004) *The State of Soils in England and Wales*. Environment Agency, Bristol.

FAOStat (2009) Available at: http://faostat.fao.org/

Fox, N.J., White, P.C., McClean, C.J., Marion, G., Evans, A. & Hutchings, M.R. (2011) Predicting impacts of climate change on *Fasciola hepatica* risk. *PLoS One*, **6**, e16126.

Gabriel, D., Thies, C. & Tscharntke, T. (2005) Local diversity of arable weeds increases with landscape complexity. *Perspectives in Plant Ecology, Evolution and Systematics*, **7**, 85–93.

Gallai, N., Salles, J-M., Settele, J. & Vaissière, B.E. (2009) Economic valuation of the vulnerability of world agriculture confronted by pollinator declines. *Ecological Economics*, **68**, 810-821.

Garibaldi,, L.A., Steffan-Dewenter, I., Winfree, R., Aizen, M.A., Bommarco, R., Cunningham, S.A. *et al.* (2013) Wild pollinators enhance fruit set of crops regardless of honey bee abundance *Science Published online 28 February 2013 [DOI:10.1126/science.1230200]*

Gibbens N. (2012) Schmallenberg virus: a novel viral disease in northern Europe. *Veterinary Record,* **170**, 58.

Gibson, R.H., Nelson, I.L., Hopkins, G.W., Hamlett, B.J. & Memmott, J. (2006) Pollinator webs, plant communities and the conservation of rare plants: arable weeds as a case study. *Journal of Applied Ecology*, **43**, 246-257.

Goulson, D. (2007) Bumblebees and other pollinating insects. *The Farm Wildlife Handbook* (ed R Winspear) pp. 105-112. RSPB, Sandy.

Goulson, D., Lye, G.C. & Darvill, B. (2008) Decline and conservation of Bumble bees. *Annual Review of Entomology*, **53**, 191-208.

Government Office for Science (2011) *Foresight. The Future of Food and Farming: Challenges and Choices for Global Sustainability*. Final Project Report. The Government Office for Science, London.

Guis, H., Caminade, C., Calvete, C., Morse, A. P., Tran, A. & Baylis, M. (2012) Modelling the effects of past and future climate on the risk of bluetongue emergence in Europe. *Journal of the Royal Society Interface*, **9**, 339-350.

Hawes, C., Haughton, A.J., Bohan, D.A. & Squire, G.R. (2009) Functional approaches for assessing plant and invertebrate abundance patterns in arable systems. *Basic and Applied Ecology*, **10**, 34-47.

Hawes, C., Haughton, A.J., Osborne, J.L., Roy, D.B., Clark, S.J., Perry, J.N., *et al.* (2003) Responses of plant and invertebrate trophic groups to contrasting herbicide regimes in the Farm Scale Evaluations of genetically-modified herbicide-tolerant crops. *Philosophical Transactions of the Royal Society of London B-Biological Sciences*, **358**, 1899-1913.

Hawes, C., Squire, G.R., Hallett, P.D., Watson, C.A. & Young, M.W. (2010) Arable plant communities as indicators of farming practice. *Agriculture, Ecosystems and Environment*, **138**, 17-26.

He, J-Z., Ge, Y., Xu, Z. & Chen, C. (2009) Linking soil bacterial diversity to ecosystem multifunctionality using backward-elimination boosted trees analysis. *Journal of Soils and SedimentsSediments*, **9**, 547-554.

Hector, A., Dobson, K., Minns, A., Bazeley-White, E. & Lawton, J.H. (2001) Community diversity and invasion resistance: an experimental test in a grassland ecosystem and a review of comparable studies. *Ecological Research*, **16**, 819-831.

Holland, J. P., Morgan-Davies, C., Waterhouse, T., Thomson, S., Midgley, A. & Barnes, A. (2011) *An Analysis of the Impact on the Natural Heritage of the Decline in Hill Farming in Scotland.* Scottish Natural Heritage Commissioned Report No. 454. SNH, Edinburgh.

Jeffery, S. & Gardi, C. (2010) Soil biodiversity under threat - a review. *Acta Societatis Zopoligicae Bohemicae*, **74**, 7-12.

Jenny, H. (1941) *Factors of Soil Formation*. McGraw Hill, New York.

Kenyon, F., Sargison, N.D., Skuce, P.J. & Jackson, F. (2009) Sheep helminth parasitic disease in south eastern Scotland arising as a possible consequence of climate change. *Veterinary Parasitology*, **163**, 293-297.

Kevan, P.G. & Phillips, T.P. (2001) The economic impacts of pollinator declines: an approach to assessing the consequences. *Conservation Ecology*, **5**, 8.

Kiær L.P., Skovgaard I. M. & Østergård H. (2012). Effects of inter-varietal diversity, biotic stresses and environmental productivity on grain yield of spring barley variety mixtures. *Euphytica*, **185**, 123–138.

Klaus V.H., Kleinebecker, T., Hölzel, N., Blüthgen, N., Boch, S., Müllerd, J. *et al.* (2011) Nutrient concentrations and fibre contents of plant community biomass reflect species richness patterns along a broad range of land-use intensities among agricultural grasslands. *Perspectives in Plant Ecology, Evolution and Systematics*, **13**, 287-295.

Kremen, C., Williams, N.M. & Thorp, R.W. (2002) Crop pollination from native bees at risk from agricultural intensification. *Proceedings of the National Academy of Science of the USA*, **99**, 16812-16816.

Lal, R. (2008) Carbon sequestration. *Philosophical Transactions of the Royal Society B-Biological Sciences*, **363**, 815–830.

Loreau, M. (1998) Biodiversity and ecosystem functioning: a mechanistic model. *Proceedings of the National Academy of Science of the USA*, **95**, 5632-5636.

Marshall, E.J.P., Brown, V.K., Boatman, N.D., Lutman, P.J.W., Squire, G.R., & Ward, L.K. (2003) The role of weeds in supporting biological diversity within crop fields. *Weed Research*, **43**, 77-89.

McCann, C.M., Baylis, M. & Williams, D.J. (2010) The development of linear regression models using environmental variables to explain the spatial distribution of *Fasciola hepatica* infection in dairy herds in England and Wales. *International Journal of Parasitology*, **40**, 1021-1028.

Morgan-Davies, C., Waterhouse, A., Pollock, M.L. & Holland, J.P. (2008.) Integrating hill sheep production and newly established native woodland: achieving sustainability through multiple land use in Scotland. *International Journal of Agricultural Sustainability*, **6**, 133-147.

Newton, A.C., Begg, G.S. & Swanston, J.S. (2009) Deployment of diversity for enhanced crop function. *Annals of Applied Biology*, **154**, 309-322.

Nichols, G., Lane, C., Asgari, N., Verlander, N.Q. & Charlett, A. (2009) Rainfall and outbreaks of drinking water related disease and in England and Wales. *Journal of Water Health*, **7**, 1-8.

Norris, R.F. & Kogan, M. (2000) Interactions between weeds, arthropod pests, and their natural enemies in managed ecosystems. *Weed Science*, **48**, 94–158.

Pimm, S.L., Russell, G.J., Gittleman, J.L. & Brooks, T.M. (1995) The future of biodiversity. *Science*, **269**, 347–350.

Powlson, D.S., Gregory, P.J., Whalley, W.R., Quinton, J.N., Hopkins, D.W., Whitmore, A.P., *et al.* (2011) Soil management in relation to sustainable agriculture and ecosystem services. *Food Policy*, **36**, 572-587.

Preston, C.D., van der Wal, R., Welch, D., Roy, D.B. & Hill, M.O. (2006) Scottish trends in vascular plants (2005). *Scottish Natural Heritage Commissioned Report No. 169*. SNH, Edinburgh.

Pritchard, G.C., Forbes, A.B., Williams, D.J., Salimi-Bejestani, M.R. & Daniel, R.G. (2005) Emergence of fasciolosis in cattle in East Anglia. *Veterinary Record*, **157**, 578-582.

Quinton, J.N., Govers, G., Van Ooost, K.V. & Bardgett, R.D. (2010) The impact of agricultural soil erosion on biogeochemical sampling. *Nature Geoscience*, **3**, 311–314.

Rathcke & Jules (1993) Habitat fragmentation and plant-pollinator interactions. *Current Science*, **65**, 273-277

Renwick, A. & Waterhouse, T. (2008) *Farming's retreat from the Hills*. Scottish Agricultural College, Edinburgh.

Scherer-Lorenzen, M., Palmborg, C., Prinz, A. & Schulze, E-D. (2003) The role of plant diversity and composition for nitrate leaching in grasslands. *Ecology*, **84**, 1539-1552.

Scottish Food Forum (2010) *Food affordability, access and security: their implications for Scotland's food policy.* http://www.scotland.gov.uk/Publications/2009/06/25143814/10

Simm, G., Villanueva, B., Sinclair, K.D. & Townsend, S. (eds.) (2004) *Farm Animal Genetic Resources*. BSAS Publication No. 30. Nottingham University Press, Nottingham.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., *et al.* (2008) Greenhouse gas mitigation in agriculture. *Philosophical Transactions of the Royal Society B – Biological Sciences*, **363**, 789-813.

Sutton, M.A., Howard, C.M., Erisman, J.W., Billen, G., Bleeker, A., Grennfelt, P. *et al.* (2011) *The European Nitrogen Assessment*. Cambridge University Press, Cambridge.

Sylvain, Z.A. & Wall, D.H. (2011) Linking soil biodiversity and vegetation: Implications for a changing planet. *American Journal of Botany*, **98**, 517–527.

Tautz, J. & Rostás, M. (2008) Honeybee buzz attenuates plant damage by caterpillars. *Current Biology*, **18**, R1125-R1126.

Taylor, R.L., Maxwell, B.D. & Boik, R.J. (2006) Indirect effects of herbicides on bird food resources and beneficial arthropods. *Agriculture Ecosystems and Environment* **116**, 157-164.

The Scottish Government (2008) *Choosing the Right Ingredients: the future for food in Scotland - discussion paper*. Scottish Government, Edinburgh.

The Scottish Government (2009a) *Recipes for Success - Scotland's National Food and Drink Policy*. Scottish Government, Edinburgh.

The Scottish Government (2009b) *Food Security: the role for the Scottish Government in ensuring continuity of food supply to and within Scotland and access to affordable food*. Scottish Government, Edinburgh.

The Scottish Government (2009c) *Food Affordability, Access and Security: their implications for Scotland's food policy*. Scottish Government, Edinburgh.

The Scottish Government (2012a) *Agriculture Facts and Figures 2012*. Scottish Government, Edinburgh.

The Scottish Government (2012b) *Abstract of Scottish Agricultural statistics 1982 to 2012*. Scottish Government, Edinburgh.

The Scottish Government (2012c) *Economic Report on Scottish Agriculture 2012 Edition*. Scottish Government, Edinburgh.

Thomson, S. (2011) *Response from the hills: Business as usual or a turning point? - An update of "Retreat from the Hills"*. SAC Rural Policy Centre, Edinburgh.

Tilman, D., Hill, J. & Lehman, C. (2006) Carbon-negative biofuels from low-input high-diversity grassland biomass. *Science*, **314**, 1598-1600.

Usher, M.B., Sier, A.R.J., Hornung, M. & Millard, P. (2006) Understanding biological diversity in soil: The UK's Soil Biodiversity Research Programme. *Applied Soil Ecology,* **33**, 101–113.

Verbruggen, E., Kiers, E.T., Bakelaar, P.N.C., Roling, W.F.M. & van der Heijden, M.G.A. (2012) Provision of contrasting ecosystem services by soil communities from different agricultural fields. *Plant and Soil*, **350**, 43-55.

Wäckers F.L., van Rijn P.C.J. & Heimpel G.E. (2008) Honeydew as a food source for natural enemies: making the best of a bad meal? *Biological Control*, **45**, 176–184

Wäckers, F.L. & van Rijn, P.C.J. (2012) Pick and mix: selecting flowering plants to meet the requirements of target biological control insects. *Biodiversity and Insect Pests: Key Issues for Sustainable Management* (eds G.M. Gurr, S.D. Wratten, W.E. Snyder & D.M.Y. Read) pp. 139-165. John Wiley and Sons Ltd, Chichester.

Williams, P., Colla, S. & Xie, Z. (2009) Bumblebee vulnerability: common correlates of winners and losers across three continents. *Conservation Biology*, **23**, 931-940.

Woodward, F.I., Bardgett, R.D., Raven, J.A. & Hetherington, A.M. (2009) Biological approaches to global environment change mitigation and remediation. *Current Biology*, **19**, R615–R623.

Chapter 4: Halting Biodiversity Loss

4.1 Summary

The ecosystem services prioritised by the Ecosystem Assessment Working Group workshop for this broad policy goal are wild species diversity (as both a cultural service and provisioning service), disease and pest regulation (regulating service), and crops (provisioning service).

With respect to wild species diversity as a cultural service:

- In upland systems, because of their less intensive management, wild species diversity is more likely to be directly regulated by biophysical and natural biotic processes than in lowland systems.
- Defining the ‘goods’ delivered by cultural services is complex. However, it is important to be clear about identifying the ‘good’ within its context, as this is likely to differ substantially between different stakeholder groups, and the delivery of different ‘goods’ may have different relationships with biodiversity and biophysical processes.
- The concept of the “appropriateness” of levels of biodiversity (e.g. raw numbers of species, or absolute values for genetic diversity) is important with respect to delivering this broad policy goal, and this will differ between the lowlands and uplands.
- Accumulating across the different cultural service goods, all species groups and levels of biodiversity are likely to play a role in regulating wild species diversity as a cultural service.

With respect to wild species diversity as a provisioning service:

- Increased biodiversity is likely to be important in terms of delivering successful ecological restoration projects and the strength of this relationship – although probably still positive - is weaker for the delivery of material for bioprospecting than for ecological restoration.
- Increased diversity overall is likely to be beneficial for harvestable species, with the exception of the diversity of certain species groups (pests and pathogens).
- Some particular species groups (vascular plants, birds, fungi, fish, and some wild mammals such as red deer and other game species) are also clearly of greater importance in terms of delivering ‘goods’ related to this service as these groups contain the harvestable species.

With respect to disease and pest regulation

- The influence of biodiversity on disease and pest regulation operates within the overall limitations imposed on biotic processes by the abiotic environment.
- The relationships between biodiversity/biophysical processes and disease and pest regulation are complex, not least because either side of the pathogen/pest – host relationship may be affected, and both are influenced by spatio-temporal variation in habitats, disturbance factors, climate and land use/management practices.
- We have some knowledge of these relationships from crop and livestock production systems, but our knowledge is poorer for the more complex natural and semi-natural systems. Such knowledge will be critical with respect to halting biodiversity loss.
- There is considerable potential for extending techniques which have been developed in production systems to explore these relationships in semi-natural systems. In addition, further detailed studies exploring the mechanisms underlying the control of disease and pest regulation are needed in all systems.

With respect to crop systems

- Intensification clearly leads to negative biodiversity impacts, with implications not only for biodiversity conservation but also for a wide range of other ecosystem services and broad policy goals.

- Sustainable farming practices will be beneficial for biodiversity in crop production systems. This will have positive feedback effects, in that enhanced biodiversity will help replace some of the services currently delivered through mechanisation or agro-chemicals.
- However, the extent to which enhanced biodiversity-delivered functions can offset the loss of production from intensive farming practice is unclear, and this will depend on the improvement in delivery of a wider range of complimentary ecosystem services
- Other changes in crop production systems, beyond simply reducing the intensity of management, may have beneficial impacts for farmland biodiversity and hence biodiversity conservation.

Overall:

- In all systems it is important to understand which elements of biodiversity are critical for delivering the aims of the broad policy goal, and how these relate to the desires of, and management by, different stakeholder groups. This level of detail is necessary for developing integrated management practices that promote biodiversity conservation.
- One of the major ways in which biodiversity can help deliver the goal of halting biodiversity loss is through its beneficial effect on a wide range of ecosystem services, particularly those which deliver an obvious 'good'. Evidence that biodiversity supports service delivery lends weight to the argument that biodiversity conservation is important beyond its own intrinsic value.

4.2 Definition of halting biodiversity loss

The policy goal of Halting Biodiversity Loss gained traction internationally through the *Convention on Biological Diversity*, which delivered important subsidiary mechanisms, e.g. the *European Habitats Directive* and the *UK Biodiversity Action Plan*. A motivating force for these major initiatives was biodiversity's inherent worth and its continued loss, with conservation to be achieved through 'sustainable development'. Biodiversity was defined as the diversity within species, of species, and between habitats and ecosystems (Ch. 1), and the aim of halting biodiversity loss implicitly refers to the conservation of all of these subcomponents.

More recently the goal of halting biodiversity loss has been given a utilitarian rationale with the rise of the ecosystem service framework and the Ecosystem Approach (Howard *et al.* 2011). Within this new context sustainable development remains an important route to delivery. Accordingly, the *Land Use Strategy* sets out key principles for sustainable land use "which reflect Government policies on the priorities which should influence land use choices." At the heart of sustainable land use is the protection of natural capital and - implicitly - the conservation of biodiversity. The *Land Use Strategy* also states that "land use decisions should be informed by an understanding of the functioning of the ecosystems which they affect in order to maintain the benefits of the ecosystem services which they provide."

Notwithstanding alternative understandings of the term 'sustainable' (POST 2012), the aims of numerous policies have now been built around the concept of sustainable development, implicitly incorporating the goal of halting biodiversity loss. Moreover, all public bodies in Scotland have a legal duty (the Biodiversity Duty, *Nature Conservation (Scotland) Act* 2004) to protect wildlife, biodiversity and natural habitats, while some key strategies address the specific goal of halting biodiversity loss. Of the latter, the most obvious is the *Scottish Biodiversity Strategy* (SBS), but other strategies are clearly relevant in terms of containing active steps to promote biodiversity conservation, including the forestry, marine, and soils strategies.

The overall aim of the *Scottish Biodiversity Strategy* "is to conserve biodiversity for the health, enjoyment and wellbeing of the people of Scotland now and in the future". The *Scottish Biodiversity Strategy* has been reviewed to address the challenges and targets for 2020 set out in the Aichi targets and the *European Biodiversity Strategy*. Although a renewed emphasis within the *Strategy* is orientated toward the utilitarian approach to biodiversity, biodiversity conservation and halting biodiversity loss are still at the heart of its ambition. .

4.3 Prioritised ecosystem services

Mace *et al.* (2012) consider the multi-layered relationship between the ecosystem service framework and biodiversity. Within this framework biodiversity can be a regulator of ecosystem processes, a final ecosystem service, or a 'good'. Successful conservation of biodiversity will influence all levels of the framework. But although biodiversity can be strongly influenced by management in relation to many different services, it is not considered a service in its own right. The conservation of biodiversity will therefore depend upon the manner in which other services are delivered.

With respect to the broad policy goal of halting biodiversity loss, particular ecosystem services may be relevant because they have either a beneficial or negative effect on biodiversity. The top four prioritised ecosystem services (from the EAWG workshop) with respect to the broad policy goal 'Halting Biodiversity Loss' are wild species diversity as a cultural service, wild species diversity as a provisioning service, disease and pest regulation (regulating service), and crops (provisioning service).

Wild species diversity is treated in the UK NEA as diversity at the species-scale (not including diversity among populations and habitats); it is also treated in a broadly cross-cutting manner, appearing as a supporting service, and as both a provisioning and cultural final service (See Table 2.2 and Fig. 2.3 of the UK NEA).

4.3.1 Wild species diversity - cultural service

With respect to wild species diversity as a cultural service, the 'goods' associated with the service are interpreted by the UK NEA in a variety of ways. In an overview of the concepts and methodological approaches (UK NEA Ch. 2), the example of a cultural service 'good' from wild species diversity is recreation (e.g. bird-watching). This kind of 'good' has clearly an element of "environmental settings" (UK NEA Ch. 16), although wild species diversity is kept separate from environmental settings within the original UK NEA framework (UK NEA Ch.2). We also consider species that are hunted as part of the recreation 'good' arising from wild species diversity.

The delivery of rare species can be interpreted as an additional cultural service 'good'. There is limited discussion within UK NEA Ch. 2 of biodiversity conservation *per se* (i.e. irrespective of the rarity or recreational value of the organisms involved). It is however commonly discussed in other chapters (for example the Broad Habitats chapters) as either a 'good' or service provided by wild species diversity, and we interpret it as being separate from the recreational 'good'. We suggest that this conservation 'good' is also likely to have been a consideration for some stakeholders during the EAWG ecosystem service prioritisation process.

Here, therefore, we consider two aspects of the cultural ecosystem service 'wild species diversity' and its relationship to biodiversity and underlying biophysical processes:

1. Wild species diversity as a cultural service in relation to recreation.

2. Wild species diversity as a cultural service in relation to biodiversity conservation.

The distinction between the two is cautiously applied. Certain charismatic species may provide a recreational service (e.g. whale-watching), as well as being a focus for biodiversity conservation. Nevertheless, it is important to attempt a distinction among these categories, as the elements of wild species diversity relevant to these two types of 'good' are not always the same, and consequently the relationship to underlying biodiversity or biological/biophysical processes may differ. Even within the second category, different elements of biodiversity may be important in different contexts: sometimes it is overall species diversity that is the ‘good’, whilst in other scenarios particular rare or iconic species are the focus.

The particular ‘good’ delivered is likely to differ between Scottish upland and lowland environments for two main reasons. First, the uplands and lowlands have different assemblages of organisms. Consequently, there may be social expectations as to which species may be enjoyed and should be conserved within these environments. Second, there are likely to be different levels of species richness in these habitats; we expect, for example, the vascular plant flora to be overall richer in lowland systems than in upland systems. Mace *et al.* (2012) point out that with respect to biodiversity conservation as a ‘good’ “One cannot… assume that high biodiversity is always the goal of conservation”. High species richness in the uplands can be seen as an indication of degraded habitat (Britton *et al.* 2009). The cultural ecosystem service of wild species diversity must therefore deliver both the species (rare and iconic) and overall level of diversity which is “appropriate” to a habitat (recognising of course that this is an artificial ideal based upon our own expectations).

4.3.2 Wild species diversity - provisioning service

As a provisioning service, wild species diversity provides the ‘goods’ of material for ecological restoration, resources for bioprospecting (where ‘latent goods’ include new medicines or compounds from natural raw materials), and food.

The UK NEA defines a number of ways in which people may value biodiversity and ecosystems (for example Box 2.1, p.19 of the Technical Report; UK NEA Ch. 2). Biodiversity is valued by many on the basis of its intrinsic value alone, but arguments based on intrinsic value have consistently failed in the policy arena (EEA 2005). Instrumental or extrinsic values associated with provisioning service ‘goods’ can be quantified as part of a service’s Total Economic Value (UK NEA Ch. 2), and lend a different and perhaps more influential weight to arguments in favour of biodiversity conservation. These additional considerations may have been important in the prioritisation of this service.

With respect to delivering the broad policy goal of halting biodiversity loss, we suggest that the ‘good’ which is of most *direct* relevance is material for ecological restoration. In this case the ‘good’ includes the appropriate target species. As species restoration programmes are commonly hampered by restricted genetic diversity in the source material (Brooker *et al.* 2011) the ‘good’ also includes genetic diversity within a species.

Whilst ecological restoration has been described as “much-needed industry worldwide” (Beattie *et al.* 2005) this service may have been prioritised during the EAWG process because other – perhaps more widely or readily-valued - ‘goods’ add weight to the argument that biodiversity conservation is beneficial to society. This includes in particular the provision of material for bioprospecting. Bioprospecting and associated activities have been the focus of a number of recent ecosystem service reviews. As stated by the EASAC (2009) review “biodiversity is the fundamental resource for bioprospecting”, and the maintenance of species diversity provides the raw material for breeding programmes. However, the ‘good’ is not necessarily the species, because modern GM techniques can isolate the gene from the species. The gene pool is enhanced by both species richness, and increased genetic diversity within a species. The service of wild species diversity could therefore deliver this ‘good’ by delivering both species diversity and within-species genetic diversity.

If we compare the delivery of provisioning services in upland and lowland habitats, with respect to bioprospecting, and including prospecting for material for ecological restoration, the ‘good’ is likely to be the rare or iconic species associated with a particular habitat. For genetic bioprospecting, in general high species and genetic diversity are critical. However, there is a tension here in that enhanced local species pools may lead to the loss of species and genes from the global species pool.

This may be particularly true in upland systems, or with the encroachment of non-native invasives in lowland systems; we return to this point in the section on ecosystem service interactions. Finally in terms of food, as the harvested items will obviously differ between upland and lowland habitats, this provisioning service must support the appropriate species for harvesting.

It is worth noting that no distinction is made between wild species diversity as a cultural and as a provisioning service in some of the analyses presented in the UK NEA. For example, Table 4.2 of Ch. 4 of the UK NEA summarises an assessment of the "importance of different biodiversity groups in underpinning the final ecosystem services". In this summary table wild species diversity is not separated into cultural and provisioning services. But the work of UK NEA Ch.4 follows the assessment's Conceptual Framework (UK NEA Ch.2). It is, therefore, implicit that in its analysis the underpinning relationships between the biodiversity groups and wild species diversity are considered similar for wild species diversity both as a cultural and a provisioning service.

4.3.3 Disease and pest regulation

Disease and pest regulation is a regulating service critical to the delivery of several final ecosystem services.

We have already discussed (Ch. 2 and 3) how the influence of biodiversity or biological/biophysical processes on the delivery of 'goods' from timber, crop and livestock production is often indirectly mediated through the occurrence and regulation of diseases and pests, either native or non-native. In the specific policy context of halting biodiversity loss, however, the 'good' is not the food or timber from farming or forestry systems.

Invasive plant and animal diseases are one of five drivers of change whose impact on UK ecosystems over the last 60 years was reviewed by the UK NEA (Ch. 3). This review concluded that pests and diseases were, overall, relatively unimportant drivers to date when compared to land use change and pollution. But in some specific cases they can lead to the loss of specific ecotypes, species or habitats from either particular regions, or (occasionally) from the entire species pool. A good example is the almost complete loss of the red squirrel (*Sciurus vulgaris*) from most of central and southern England due to competition from, and disease carried by, North American grey squirrel (*Sciurus carolinensis*; Carroll *et al.* 2009).

Enhanced pest or disease control might not be the only 'good' resulting from the service of disease and pest regulation that is relevant to halting biodiversity loss. In some cases diseases and pests may be important in keeping potentially dominant or invasive species under control. For example the dominance of marram grass in European coastal habitats is limited by the build-up of soil pathogens; in the absence of these pathogens, for example in southern-hemisphere systems where marram grass has been introduced, the marram grass becomes dominant and normal successional processes are restricted (Van der Putten *et al*. 1993). One of the major current concerns with respect to species translocations is the possibility of moving an organism to a new environment, thereby freeing it from natural predators, pests and diseases such that it is capable of rapid population expansion to the detriment of the recipient habitat or ecosystem (Brooker *et al.* 2011). This potential escape from natural enemies is the basis of the "enemy release hypothesis" in invasive species ecology (Callaway & Maron 2006). The role played here by disease and pest regulation is to maintain an adequate check on potentially-dominant species. In this sense, then, disease and pest regulation is acting as a supporting service, rather than a final (regulating) service which delivers a 'good'. However, we do not focus on this supporting service aspect of disease and pest regulation. We focus instead on the 'goods' delivered by disease and pest regulation as a final regulating service, i.e. disease and pest control (as shown in Table 2.2 of the UK NEA, reproduced in Ch. 1 of this document).

In terms of the contrast between upland and lowland systems, variation in the 'good' delivered is presumably related to the types of pests and diseases associated with the species that inhabit these environments. So for example in the uplands, the 'good' might be reduced incidence of tick transmitted Lyme disease, whilst in the lowlands it might be reduced incidence of *Phytophthora* or of aphid pests and associated plant viruses.

4.3.4 Crops

'Crops' is consistently treated as a Provisioning Service in the UK NEA, and throughout the NEA the term 'crops' is generally used to refer to those types of annual arable or horticultural crops associated with enclosed farmland, and which produce products for human consumption.

The Enclosed Farmland chapter of the UK NEA (Ch. 7) also follows this general definition, but broadens it out to include perennial crops and biomass crops (used to support energy production for human usage). Note that the term 'crop' is also used in the UK NEA Woodland chapter (Ch. 8) when considering the management of woodland and forestry to provide timber products for human use. Here, however, we assume that it was the definition in an agricultural context that was applied when this ecosystem service was prioritised with respect to the broad policy goal of halting biodiversity loss. Hence our consideration of crops focuses on lowland agricultural production from both an arable and intensively-managed grasslands perspective (consistent with a similar use of the term in Ch. 2 and Ch. 3 of this review).

The 'good' delivered by the ES of crops is generally some kind of plant matter. This can be a food product for direct human or animal consumption; alternatively it might be a crop such as linseed which is grown not for food production but as the raw material for a manufacturing process. However, within the context of the broad policy goal of halting biodiversity loss, and as in the case of disease and pest regulation, the key 'good' may in fact be the avoidance of a dis-benefit (i.e. a 'bad').

Over the last decade, the UK has produced more food per year from crops than at any other time in history. Large increases in the productivity of all crops occurred between 1940 and 2008, as exemplified by average UK wheat yields which increased from 2.5 t ha^{-1} (tonnes/hectare) to 8 t ha^{-1}. These increases in productivity have been driven by targeted varietal improvements through plant breeding, increasing chemical inputs (e.g. more effective pesticides, increased inorganic and organic fertiliser use) and mechanisation and other technological improvements (UK NEA Ch. 15). However, such increases in yield have had a clear environmental cost. In Ch. 2 we discussed the potential negative impacts of intensive farming on the delivery of a low carbon economy. Agricultural activity (intensification) has also been highlighted as one of the main drivers for the decline in biodiversity (at all levels) in the UK, Europe, and globally (Henle *et al.* 2008; Tscharntke *et al.* 2012). Sustainable farming practice is focussed on balancing the delivery of food against the negative impacts of farming on the environment. In many areas halting biodiversity loss can only be achieved by more sustainable farming, and the avoidance of the dis-benefit of declining farmland biodiversity (UK NEA Ch. 7). Therefore, in the context of this broad policy goal, all of the 'goods' provided by the ecosystem service of crops are not of equal relevance. What is critical is the delivery of crop 'goods' which are associated with a reduced impact on the natural environment. This is analogous - to some extent - to the focus on crop 'goods' from low input farming systems considered in Ch. 2 of this review, although additional factors need also to be taken into account (as discussed in more detail, below).

Although Scotland's agricultural land is dominated by upland rough grazings, the lowland agricultural systems based on arable and intensively managed grassland production fuel a major component of Scotland's annual agricultural production. In terms of how this good might differ between upland and lowland systems, crop production (as defined here) is clearly focussed very strongly in lowland environments. About 11% of Scotland's land area is arable agriculture, mainly on lower ground (<300

m) in the east of the country. Improved grasslands are also an important feature of Scotland's agricultural systems and the value of improved grassland for grazing livestock and the productivity of grasses grown under Scottish environmental conditions has led to the creation of extensive areas of improved grassland pastures across much of lowland Scotland (UK NEA Ch. 19, Figure 19.14). Hence the discussion of this ecosystem service, its resulting 'goods', and their underpinning by biodiversity and biotic/biophysical processes, is very much restricted to lowland systems in Scotland.

4.4 How do biodiversity and biotic and biophysical processes underpin these services and goods?

4.4.1 Wild species diversity – cultural service

Large scale variation in wild species diversity in Scotland can be explained as a consequence of biophysical heterogeneity which emerges as the combined effect of multiple environmental/resource gradients. First, there is a substantial climatic gradient from the relatively oceanic west coast to the relatively continental east coast (Fraser Darling & Morton Boyd 1964). This climatic gradient substantially affects species ranges with, for example, 'temperate rainforest' communities restricted to the west coast, through to sub-arctic (montane) or sub-boreal (forest) systems in the eastern watersheds (Hill & Preston 1998; Ellis *et al*. 2007; Preston *et al*. 2011). Second, Scotland's mountainous topography imposes local microclimatic variation on the larger-scale macroclimatic contrasts, with varied rock and soil types generating additional environmental heterogeneity, which is expected to increase species richness (cf. Lundholm 2009). Together this amalgam of environmental conditions creates the framework for biodiversity patterns, providing opportunities for species with widely contrasting ecological niches, and playing an important role in explaining the type and richness of species found.

However, these biophysical processes do not always promote *diversity* in an absolute sense. The low temperatures and high precipitation found in Scottish mountains restrict the diversity of some species groups (Brooker 2011). Very low resource availability can propagate through food chains, limiting the occurrence of higher trophic groups (Post 2002). Biophysical processes therefore shape the baseline condition for wild species diversity, and this baseline is an important part of delivering the 'goods' for recreation or conservation. Superimposed on the biophysical framework, there is an interesting role for historic precedent and cultural perception in determining baselines and what is deemed appropriate (see, for example, Brooker 2011). Key drivers of environmental change such as climate change or N deposition influence the regulatory biophysical underpinning, moving systems away from their "expected" baseline state and thus threatening the delivery of the service.

The UK NEA considered the relative importance of separate species groups in delivering wild species diversity. This indicated that virtually all species groups were considered to be highly important, with the exception of those groups for which biophysical conditions were generally limiting: fish and amphibians (medium importance) and reptiles (low importance). Generally the level of agreement and consensus and the availability of evidence for this assessment are medium to high (UK NEA Table 4.2). The appendices to Ch. 4 of the UK NEA include more detailed evidence of the role of these separate species-groups with respect to supporting wild species diversity, giving additional detail with respect to recreation and conservation (as summarised in Appendix 2 of this report). In most cases the described relationship is a positive contribution from each group to overall diversity simply through its existence, and sometimes - as in the case of keystone, pioneer, foundation, fodder, or prey species - enabling other species to occur. A smaller number of taxa are singled out as being important specific components of recreational activities, i.e. iconic species for recreation. In both cases the biodiversity-ES relationship is positive.

There are other more complex ways in which biodiversity in the broader sense of the CBD (genetic, species and habitat diversity) can help to deliver wild species diversity's cultural service 'goods'. In terms of genetic diversity, genetic variation is widely assumed to be related to ecological fitness and evolutionary potential (Ellstrand & Elam 1993; Reed & Frankham 2003). Species conservation is therefore directly facilitated by genetic diversity. This has been emphasised by recent studies of strategies for species conservation during climate change (Hoffman & Sgrò 2011), including the goal of protecting evolutionary potential and enabling adaptation to a changing environment.

In terms of the role of species diversity, diversity in potential food sources may be essential for maintaining stable food chains and the success of the higher trophic levels which include many iconic species. Studies of domestic grazers have shown that increased fodder species diversity increases food intake and nutrition in browsers, and ultimately domestic herbivore production (Wang *et al.* 2010). Studies of invertebrate species indicate that these effects may operate in natural and semi-natural systems as well. For example, increased richness of food sources increases the fitness of invertebrate herbivores such as grasshoppers (Unsicker *et al*. 2010). These effects may be because higher species richness reduces temporal variability in resource supply: higher plant diversity provides more temporally consistent food resource availability to arthropod food webs (Haddad *et al.* 2011), and increased substrate diversity reduces variability in the composition and function of soil communities (Keith *et al.* 2008).

Habitat diversity can also regulate the species abundance and population stability. Evidence that habitat diversity is related to availability of hunting species can be found from a number of sources. For example Smith *et al.* (2005) found that low habitat diversity was associated with long-term declines in the numbers of European hares (*Lepus europaeus*). North American elk (*Cervus canadensis* or *elaphus*) when not in forested habitats look for areas of high vegetation diversity, and in particular habitats with the presence of shrubs, which have been described as providing "hiding cover requirements" (Sawyer *et al.* 2007). Higher levels of survival in red-legged partridge are associated with increasing habitat diversity, and it has been proposed that this is due to reduced mortality from both predation and disease – increasing habitat connectivity and diversity is considered important for preventing population declines in this species (Buenestado *et al.* 2009). In all these cases diversity is the diversity of different habitat types, rather than structural diversity (e.g. variation in vegetation height) within a habitat. Wild species diversity of diminutive organisms such as lichens and bryophytes may also be influenced by habitat diversity. For example epiphytic lichen diversity is in part an outcome of spatial turnover among habitat patches, with wild species diversity dependent on the landscape heterogeneity of foundation species and habitats (Ellis 2012).

Finally, habitat diversity is obviously not simply an outcome of species diversity. Certainly in the semi-natural systems dominating much of the UK management impacts are a key driver of habitat diversity. Woodcock & Pywell (2010) demonstrated the role of habitat diversity in a study of invertebrate herbivore species richness in calcareous grasslands. They showed that herbivore species richness was positively correlated with both forb and grass species richness (demonstrating again the importance of occurrence and richness of particular functional groups), and suggested that variety in the timing and type of management to promote heterogeneity in sward structure (i.e. structural diversity within a habitat) was also critical to promoting invertebrate diversity.

When contrasting the uplands and lowlands, different biophysical properties clearly underlie qualitative differences in their wild species diversity. Put simply, the 'expected' levels of biodiversity and the iconic species of conservation and recreation interest differ between these systems. The differences are clearly demonstrated in the Broad Habitat Type (BHT) chapters of the UK NEA that focus on key upland and lowland systems. For example, the Mountains, Moorlands and Heaths chapter (UK NEA Ch. 5) provides considerable discussion of the role of wild species diversity as a cultural service for recreation and conservation in the UK uplands. Charismatic and important species of mountains, moorlands and heaths include culturally significant breeds such as highland cattle, whilst iconic species such as peregrine falcon, golden eagle and nightjar are associated with the sense of place that attracts tourism. Other iconic species from these systems, for example red deer and grouse, are associated with hunting activities. In contrast, the Semi-natural Grasslands chapter (UK NEA Ch. 6) describes some examples of cultural service 'goods' associated with recreation and conservation delivered by this important lowland BHT. For example, lowland semi-natural grassland habitats contain a large number of species of conservation concern, and the Biodiversity Action Plans for many flagship conservation species require conservation of semi-natural grasslands. At the same time semi-natural grasslands are key elements of many scenic National Park areas, and contribute substantially to the "environmental setting".

Biophysical processes regulate differences in wild species diversity between upland and lowland systems not only directly, by regulating which species can survive, but also by determining the intensity of management (Fraser Darling & Morton Boyd 1964; Brooker 2011). In the uplands, low overall productivity and a harsher climate has led to low human population density and intensity of land management (Brooker 2011). Physical factors such as topography and rock type are therefore likely to have a greater absolute influence on the ecosystem service of wild species diversity in the uplands.

With respect to wider aspects of biodiversity (genetic, species and habitat diversity) there is limited information on how diversity at a genetic and species level is associated with or might underpin the occurrence of iconic species specifically in either the uplands or lowlands, although the relationships may follow the generic patterns discussed above. In terms of habitat diversity, Table 5.5 of the UK NEA (p.136) indicates that within the mountains, moorlands and heaths BHT, dwarf shrub heath and bog habitats are most important for deer and game bird provision for hunting, whilst these habitats are also important for tourism and recreation associated with watching wildlife, with the addition of upland fen, marsh and swamp. Consequently there is likely to be a positive relationship between habitat diversity and those 'goods' from wild species diversity that are associated with recreation and conservation. However, for the lowlands, and semi-natural grassland in particular, it is not clear which elements of biodiversity are vital for the delivery of this cultural ecosystem service – whether it is the rare species, or the common species, and how the occurrence of these different species groups relates to underlying diversity of genes, species or habitats.

4.4.2 Wild species diversity – provisioning service

The role of biophysical processes in regulating wild species diversity as a provisioning service is similar to that for wild species diversity as a cultural service, i.e. biophysical processes set a broader framework within which biological system/biodiversity-ES-'goods' relationships then operate.

Ecological restoration is being considered as a possible strategy for enhancing the provision of ecosystem services. This includes enhancement of wild species diversity through the prevention of biodiversity loss (Bullock *et al.* 2011). For the provision of material for ecological restoration, greater genetic diversity within species at donor sites will have a positive effect on the availability of suitable material. More genetically diverse samples help with the avoidance of inbreeding depression in *ex situ* collections or in newly established populations in the wild (Menges 2008). However, increased overall species diversity within a habitat might not lead to the enhanced availability of the target species, particularly if this is associated with an influx of non-native species. Therefore, the link between biodiversity and delivery of material for ecological restoration might not always be positive.

It could be assumed that more species-rich systems would be better in terms of providing material for bioprospecting. Although the potential for bioprospecting is often discussed in the context of species-rich ecosystems such as tropical rainforests, many compounds have been derived from a wide range of environments across the globe. As a number of reviews (e.g. Beattie *et al.* 2005; EASAC 2009) observe, it is virtually impossible to predict which ecosystems or species will become an important bioprospecting resource. On this basis we could propose that greater biodiversity in all environments will better deliver materials for bioprospecting. But Tulp & Bohlin (2002) suggest that "important molecular mechanisms are likely to be ubiquitous", and hence there are no obvious advantages to wide-scale biodiversity prospecting. From this we might conclude that there is substantial functional redundancy, and no benefit from increasing biodiversity for the provision of raw material for bioprospecting. However, this fails to assess how close we currently are to having covered all of the important molecular mechanisms. In addition, in some limited cases there does seem to be a level of predictability in terms of how biological systems underpin bioprospecting opportunities. Some types of “bioactive products” are non-randomly distributed within species groups. The origins of many drugs are from a relatively restricted range of well-known taxa (Zhu *et al.* 2011). Areas with high herbivore densities may also be important for bioprospecting, as anti-herbivore defence compounds are often the basis for pharmaceutical derivatives (Beattie *et al.* 2005). Extreme (and often species-poor) environments (e.g. thermal vents, arctic vegetation, and mine waste) have also yielded organisms with exploitable and important biomolecules (Beattie *et al.* 2005). In these latter cases the critical discovery process is often a targeted search in a relevant system for a particular product (e.g. heat-tolerant enzymes in species from hot springs), but this search is still promoted by increasing biodiversity within these target systems, and hence by the conservation of biodiversity. The case for promoting conservation of biodiversity in order to support bioprospecting potential is further strengthened when we consider that “a high proportion of commercially important species are either small or microscopic, and so losses go undetected” (Beattie *et al.* 2005, 2011).

As stated by Mace *et al.* (2012) “the potential value of wild relatives and the potential benefits from bioprospecting increase directly with the number and evolutionary distinctiveness of species”. Consequently, and in so far as the delivery of bioprospecting and genetic diversity ‘goods’ are influenced by factors that regulate biodiversity overall, the diversity of landscape units will also influence the delivery of these ecosystem services, and excessive fragmentation of habitats will lead to reduced diversity (Diaz *et al.* 2005). There is, therefore, a need to preserve within-species (i.e. genetic), species, and habitat/landscape diversity in order to best promote wild species diversity as a provisioning service for bioprospecting. Notably, although bioprospecting and the provision of genetic material as a resource for breeding programmes are perhaps the provisioning services that are most closely dependent on overall levels of biodiversity, at the same time they are perhaps the least important (in terms of value) of the provisioning services, certainly within a European context (EASAC 2009).

Finally in terms of food provision from wild species diversity, it is likely that that there is a generally positive relationship between biodiversity and provision of ‘goods’. This is firstly because some species groups clearly deliver specific goods: grazing mammals, birds, vascular plants, fungi, fish, and shellfish, for example, are particularly important for provision of wild foods. However, many of these harvested organisms are likely to themselves be dependent on their own genetic diversity (in that high within-species genetic diversity promotes population sustainability), and on species and habitat diversity. For example, Nelson *et al.* (2008) discussed 17 species that are considered important forage species for humans in boreal Alaskan habitats. These 17 species have quite different habitat requirements, and hence mosaics of habitats generated by an active fire cycle within the woodland are important in maintaining forage species availability and diversity. The role of biodiversity in regulating provisioning services is currently one of the major knowledge gaps in the field of ecosystem services (UK NEA Ch.4)

In terms of contrasting the biological/biophysical underpinning of services in upland and lowland habitats, the Mountains, Moorlands and Heaths chapter (Ch. 5) of the UK NEA provides little information on how wild species diversity as a provisioning service is underpinned by biodiversity in this particular BHT. The Semi-natural Grasslands chapter (Ch. 4) notes that many garden plants have been sourced from semi-natural grasslands, and perhaps this an indication of the potential of these systems for delivering bioprospecting goods. Alternatively this relationship may simply reflect the romantic ideal of having gardens that in some way create an idealised version of nature, with species from grassland systems being amenable to cultivation. But bioprospecting in such habitats is not limited to sourcing material for horticulture: seeds have also been sourced from them for conservation action, and rare and traditional livestock breeds are associated with semi-natural grasslands (and mountains and heaths). Overall we conclude that, as for wild species diversity as a cultural service, the closer the ecological status of the system to the conservation optimum, the better for the delivery of wild species diversity as a provisioning service.

4.4.3 Disease and pest regulation

Specific examples of how diseases and pests of production systems might be regulated by the biotic and abiotic environment are provided in Ch. 3 of this review. Here we focus more on pests and diseases in the wider environment, and in natural and semi-natural systems, and their possible influence on biodiversity conservation and halting biodiversity loss.

At a broad level, biophysical processes will regulate diseases and pests (and thus will influence delivery of disease and pest regulation) by providing or limiting the environmental conditions necessary for pest and pathogen transmission and survival. For example, a change in the distribution and incidence of liver fluke in Scotland, and the recent outbreaks of insect-borne viral diseases of livestock such as Bluetongue and Schmallenberg in the South of the UK, have been linked to both climate change and the prevailing wind direction (Willson & Mellor 2009). The impact of climate on pests and diseases might be direct, for example by limiting the occurrence of necessary conditions such as high humidity or winter days below freezing for *Phytophthora* species, or survival of parasite eggs on pasture. It might also be indirect, limiting the distribution of intermediary species on which the pests and pathogens depend (for example snails and biting midges for liver fluke and Bluetongue virus, respectively). Within a particular (abiotic + biotic) environmental context, biota influence disease and pest regulation.

Genetic diversity provides resilience to infectious disease in agriculturally important crops and livestock species. As discussed in Ch.2, crops and livestock are increasingly selected for specific production traits, and approved crop cultivars by definition reduce genetic diversity, leaving populations vulnerable to new disease threats. Wild or “less selected” populations provide a backup source of novel resistance, linking pest and disease regulation to wild species diversity as a provisioning service. But such populations, particularly rare livestock breeds, are under threat due to a lack of immediate commercial benefit, as detailed for livestock in the *UK National Action Plan on Farm Animal Genetics Resources* (Defra 2006).

In some situations the impact of maintaining genetic diversity may be negative, as in the case of pest/pathogen populations. Such “unwanted” diversity can support enhanced infection rates and provide populations resistant to disease control measures such as the anthelminthics used to control intestinal worms in livestock (Kenyon *et al*. 2009). Likewise the impact of species diversity on pest and disease regulation can be ambivalent. Increasing species diversity can limit, through competition for resources, the population size of host species (thus restricting transmission of the pest or disease), or it can provide alternative hosts (thus promoting transmission of the pest or disease; Newton *et al.* 2009, Birch *et al.* 2011). Beyond genetic, species and habitat diversity, the occurrence of certain species (functional) groups may be particularly important, for example the role of pathogenic soil organisms in preventing the dominance of particular plant species (Callaway & Maron 2006; van Grunsven *et al.* 2007) or destroying parasite eggs and larvae.

Consequently we can see that the relationship between biodiversity and pest and disease regulation is complex, and can be positive or negative, depending on the context and dominant process. Critically, we know far less about the regulation of pests and diseases and its influence on biodiversity conservation in natural Scottish systems than we do about their role in livestock or crop production systems. It is not possible to say how the influence of biodiversity on the delivery of this service might differ between upland and lowland systems in Scotland. This is despite some specific cases in Scottish semi-natural environments where pest-host interactions have been studied in detail, for example the role of intestinal parasites in regulating grouse population dynamics (e.g. Sievwright *et al.* 2004), and the role of deer management in regulating tick abundance (Gilbert *et al*. 2012). This indicates that there is substantial potential for applying the techniques developed for production systems, such as the disease surveillance and diagnostic services provided for the livestock industry, to the understanding of disease and pest regulation (and its consequences) in natural systems.

Important differences exist between production and natural systems in terms of the processes operating to regulate pests and pathogens and their impacts. These differences could make the direct transfer of information from production systems to natural and semi-natural systems difficult. First, in intensive and extensive food production systems pests and disease-causing organisms are a problem, and so the potential benefits from pests and diseases in more natural and semi-natural systems (in regulating dominant species, for example) may have been overlooked. Crop production (for example) is focussed on maximising output from single species, whereas halting biodiversity loss is about conserving all levels of biodiversity – genetic, species, and habitat diversity. Disease and pest regulation might influence habitat and genetic diversity, as well as species diversity. At the genetic level the rapid evolution of pathogens to evade protective host responses in turn drives and maintains genetic diversity in host populations, enabling the maintenance of otherwise sub-optimal (but resistant) genotypes. The extreme level of allelic diversity within the vertebrate major histocompatibility complex (MHC) is a good example of this. Large numbers of alleles are maintained within a population at loci involved in the immune response to pathogens by some form of balancing selection (Hughes & Yeager 1998). In extreme cases, this could also have the alternative effect of causing genetic bottlenecks and population fragmentation during an epidemic, although bottlenecks can be beneficial in purging deleterious alleles from the gene pool (Leberg & Firmin 2008). At the habitat level pests and diseases are important particularly when they influence what might be regarded as keystone species, for example heather beetle in heather moorland, or species which play a critical but beneficial role in the life-cycle of keystones (bees and *Varroa* mites).

Although our understanding of diseases and pest regulation might be better for production systems, it is still far from complete. For example, the biocontrol of pests is an ecosystem service that has high value for humans (Costanza *et al.* 1997, Losey & Vaughan 2006) but is under increasing challenge from human population growth, climate change and loss of functional biodiversity (Costanza *et al.* 1997; Naidoo *et al.* 2008; Porter *et al.* 2009; The Royal Society 2009). The relationship between the presence of semi-natural habitat and diversity and abundance of beneficial and pest arthropods is well described in the literature (Landis *et al.* 2000; Bianchi *et al.* 2007; Winqvist *et al.* 2011). In order to find appropriate management policies and practices that increase ecosystem services provision and benefit for people (i.e. increase yield, reduce crop losses, reduce pesticide use and increase functional biodiversity), the mechanisms that underpin this relationship, as well as the conditions for optimising ES provision need to be uncovered. This would take effect through the management of the composition, level and arrangement of semi-natural habitats identified as being high priority for maintaining biodiversity that delivers these key ecosystem services.

Overall, it is clear that there is a strong link between halting biodiversity loss and the regulation of pest and diseases. In a direct sense this is because pest and disease regulation influences our attempts to halt biodiversity loss, having both positive and negative effects for biodiversity conservation. Indirectly the possible benefits of biodiversity in the wider landscape for the regulation of pests and diseases in production systems, and to promote the maintenance of source material for breeding disease and pest resistance, acts as a powerful force in favour of halting biodiversity loss throughout landscapes (i.e. beyond the confines of protected areas). But, as concluded in the UK NEA “Understanding how to better manage ecosystems to control pests and pathogens requires detailed studies to describe host-pest and host-pathogen interactions and to understand how these alter in response to environmental changes”.

4.4.4 Crops

As in the previous sections it seems sensible to consider first the role of biophysical processes before going on to consider the role of biological processes and biodiversity in regulating the role of crops in delivering the broad policy goal of halting biodiversity loss. The natural environment of Scotland and its vegetation, landscapes and wildlife are products of a different (and for the most part longer) geological history than most of the rest of the UK, a colder and wetter climate, and a very different history of land use. Across much of Scotland the topography, climate and underlying nutrient poor nature of the soil impose severe constraints on the natural productive capacity of the land (Bibby *et al.* 1982, Brown *et al*. 2011). This of course limits crop production to particular regions within Scotland (UK NEA Ch. 19), and hence the impact of the delivery of this service on the broad policy goal of halting biodiversity loss is also restricted to lowland environments. It is within these systems that biodiversity plays a role in regulating ES delivery.

As would be expected given its importance, and hence the wide range of research directed at improving agricultural production over the years, this ES and the way it is supported by biodiversity and/or biophysical processes is given a lot of consideration across different chapters of the UK NEA. The generalities of the ES's importance at a UK and Scottish level are particularly highlighted in Chapters 7 (enclosed farmland), 15 (provisioning services) and 19 (Scottish summary).

But it is probably useful to revisit the question being asked in our review, and in particular in this Chapter. Here, we are not examining how biophysical processes and biodiversity influence crop production and yield. These are discussed in Ch.3 of this review. Instead we are considering explicitly the role of crops in delivering the broad policy goal of halting biodiversity loss. As stated above (Section 4.3.4) perhaps the most critical role of the ES of crops for this broad policy goal its potential negative impacts for biodiversity conservation. Ch.3 sets out in detail the negative impacts of crops on some components of agricultural biodiversity, for example the reduction in lowland agricultural systems of diversity in soil microbial communities (Daniell *et al.* 2001), non-crop arable vegetation (Gabriel *et al.* 2005; Hawes *et al.* 2009) and lowland grassland vegetation (Klaus *et al.* 2011). But it is possible to reduce the impact of crop production as a driver of biodiversity loss. Ch. 2 explains the way in which different elements of the biota and biodiversity can support less intensive agricultural systems, whose reduced demands for mechanical and chemical inputs help to promote a low carbon economy. The additional benefit of this move toward less intensive agricultural practice is a reduced impact of crop production on biodiversity. Because this information is set out in detail in Ch.2, and is to some extent reiterated in Ch. 3, we will not repeat it here.

Of direct relevance here, however, is that reduced intensification only goes part way in terms of the possible modifications of crop production that might help in halting biodiversity loss. Other changes in management practice may be important. For example, Batary *et al.* (2011) found that simple reductions in the intensity of management of crop systems explained only some of the response of farmland biodiversity across a range of farming systems. Firstly, responses were specific to particular elements of farmland biodiversity; for example, non-carnivore carabids and hunting spiders but not grasshoppers benefited from a change to organic farming. Secondly biodiversity responses were also dependent on the size of management unit, i.e. field size. They conclude that "The great differences in responses of functional groups to local cereal and grassland as well as landscape management suggest implementing more scale and group specific targets for agri-environmental schemes to improve their efficiency." Changes in plot scale – although likely to be beneficial in terms of halting biodiversity loss - are clearly a management decision. There is then likely to be very little influence of underlying biodiversity on such a decision, or much of a supporting role for biodiversity in delivering these changes.

Finally, as the evidence set out in Ch.2 and Ch.3 makes clear, various components of biodiversity – for example soil, pollinator, genetic, species and habitat diversity – can all help promote more sustainable crop production systems with reduced negative impacts for biodiversity conservation. Because of this supporting role for biodiversity, perhaps an additional key role of the ES of crop production in delivering the broad policy goal of halting biodiversity loss is not related to crop production itself, or its negative impacts on the environment. As for natural regulation of pests and diseases, the possible benefits to crop production from biodiversity (for example the key role of pollinators) act as a powerful argument in favour of conservation.

4.5 Interactions between ecosystem services

The clear message that different 'goods' are dependent on different components of biodiversity indicates that there are likely to be interactions between the different prioritised ecosystem services within this broad policy goal, and indeed even between the deliveries of different 'goods' from a given service.

With respect to wild species diversity as a cultural service, there are clear interactions between the 'goods' of biodiversity conservation and field sports. The focus of field sports on the promotion of a single species may have negative impacts for other recreational sectors: this may be legitimate, e.g. grouse moor management, or illegitimate, e.g. persecution of top predators (Amar *et al.* 2012; Redpath *et al.* 2013). Different stakeholder groups expect different iconic species and associated "appropriate" levels of diversity. Some of these may be mutually exclusive 'goods'. Habitats may also tend towards homogeneity at large spatial-scales, becoming artificially reduced in diversity to facilitate high population densities of a small selection of species. Thus, biodiversity can be subject to intervention in order to maximise derived cultural service goods for one particular recreational sector, and this intervention may reduce the delivery of goods to other sectors, or limit (by reducing overall biodiversity) achievement of the overall goal of halting biodiversity loss.

There may also be conflict in terms of what constitutes an "appropriate" level of biodiversity (at all scales, from gene to habitat). The hypothetical ideal for bioprospecting is as much genetic diversity as possible. However, biodiversity conservation aims to deliver the level of biodiversity that is characteristic to any given habitat, and in environments such as the Scottish uplands this might be low. Also, there might be some complex within-service interactions. It is perhaps the overall (global) pool of species that is most important for bioprospecting. Enhanced local-scale species richness due to an influx of invasives which leads to the loss of species from the global pool would not be a benefit to bioprospecting.

"Appropriateness" of levels of biodiversity is also an area of potential conflict within the delivery of disease and pest regulation. As noted, diseases and pests can have both negative and positive interactions in terms of promoting biodiversity. So how much do we need in the way of pests and pathogens in wild systems to maintain a healthy ecosystem, and how do we detect and control them? As with wild species diversity as a cultural service, pests and diseases are one area where tensions between the role of services in helping to halt biodiversity and delivering other broad policy goals may readily arise. For example, the intensification of farming practices in Europe are generally regarded to impact indirectly on farmland birds through loss of invertebrate and plant food. However, recently published evidence from the USA suggests that the use of pesticides in the USA may be having a direct toxic effect on grassland birds (Mineau & Whiteside 2013). Intensive farming practices for enhanced crop or livestock production might make disease and pests more prevalent: artificially high grazer numbers can increase parasite burdens in wild organisms, and anthelmintic resistance is now common in populations of livestock nematode parasites leading to breakdowns in parasite control (Sutherland & Leathwick 2011). Intensive farming practices (e.g. overuse of pesticides) can also reduce the abundance of natural invertebrate predators, leading to increased pest prevalence. However, actions aimed at halting biodiversity loss can also have negative impacts on e.g. production systems. For example, an increased incidence of liver fluke - which uses snails as an intermediate host – has been linked to environmental policy which encourages farmers to develop wetlands for conservation (Pritchard *et al*. 2005). This final example illustrates the inherent tension between production systems focused solely on the delivery of a particular 'good', for example the intensive production of timber or crops, and biodiversity conservation.

But not all of interactions between services are negative. Less intensive farming practice, which can help deliver a lower carbon economy and perhaps more sustainable (in the long term) food production, also has benefits for halting biodiversity loss (Bignal & McCracken 1996, 2000; Firbank *et al.* In Press). And as we have pointed out, the supporting role of biodiversity in enabling more sustainable farming, as well as in providing raw material for breeding disease and pest resistance in production systems, in turn helps to support the goal of halting biodiversity loss by promoting the idea that biodiversity is worth conserving.

4.6 Knowledge gaps

With respect to wild species diversity as a cultural service, one of the major challenges is in understanding which elements of the biota and biodiversity (genetic, species, habitat) are considered important for service delivery. In some cases, where iconic species are the focus for recreation, this is relatively straightforward. In others, where the cultural service is delivered by the wider environment (rather than obviously a single organism) it is much more complex. Developing this understanding is necessary before we can start mapping areas of conflict among contrasting cultural service goods, and learning how to prioritise certain goods where conflicts exist.

With respect to disease and pest regulation, pest and disease regulation in natural or semi-natural systems is relatively under-studied, and the potential benefits of pests and diseases for biodiversity conservation are often overlooked. Additional targeted studies of particular pest-disease-host relationships are needed to understand the specific underlying relationships and processes, and how these might be mediated by biodiversity and affected by climate change or other biotic stressors. However, there seems to be substantial potential for developing approaches from production systems, particularly in disease surveillance, molecular diagnostics and pathogen host interactions and applying them in semi-natural/natural habitats. A good example of such an application has been in understanding the relationship between the endangered native red squirrels and viruses carried by North American grey squirrels (McInnes *et al.* 2006). In addition there needs to be an exploration of the possible balance between the pros and cons of pest and diseases for halting biodiversity loss. This stems not least from the greater complexity of natural systems, and the possibility for complex indirect as well as direct effects of pests and diseases.

With respect to crops, a clear identified knowledge gap is how to alter management practices beyond simply reducing production intensity in order to benefit overall biodiversity. Other actions may be possible in addition to changing the scale of management practices, for example actively promoting certain habitat types. The role of underlying biodiversity in helping to deliver this would need to be explored.

References

Amar, A., Court, I.R., Davison, M., Downing, S., Grimshaw, T., Pickford, T. *et al.* (2012) Linking nest histories, remotely sensed land use data and wildlife crime records to explore the impact of grouse moor management on peregrine falcon populations. *Biological Conservation*, **145,** 86-94.

Batary, P., Holzschuh, A., Orci, K.M., Samu, F. & Tscharntke, T. (2012) Responses of plant, insect and spider biodiversity to local and landscape scale management intensity in cereal crops and grasslands. *Agriculture, Ecosystems and Environment*, **146**, 130- 136.

Beattie, A. J., Barthlott, W., Elisabetsky, E., Farrel, R., Kheng, C. T. & Prance, I. (2005) New products and industries from biodiversity. *Ecosystems and human well-being: Current state and trends: Findings of the Condition and Trends Working Group, Washington (D.C.)* (eds R. Hassan, R. Scholes & N. Ash), pp. 272-295. Island Press, Washington, D.C..

Beattie, A. J., Hay, M. E., Magnusson, B., de Nys, R., Smeathers, J. & Vincent, J. F. V. (2011) Ecology and bioprospecting. *Austral Ecology*, **36,** 341-356.

Bianchi, F., Honěk, A. & van der Werf, W. (2007) Changes in agricultural land use can explain population decline in a ladybeetle species in the Czech Republic: evidence from a process-based spatially explicit model. *Landscape Ecology*, **22**, 1541-1554.

Bibby, J.S., Douglas, H.A., Thomasson, A.J. & Robertson, J.S. (1982) *Land Capability Classification for Agriculture*. The Macaulay Institute for Soil Research, Aberdeen.

Birch, A., Begg, G. & Squire, R. (2011) How agro-ecological research helps to address food security under new IPM and pesticide reduction policies for global production systems. *Journal of Experimental Botany*, **62**, 3251-3261.

Bignal, E.M. & McCracken, D.I. 1996. Low-intensity farming systems in the conservation of the countryside. *Journal of Applied Ecology*, **33,** 413-424.

Bignal, E.M. & McCracken, D.I. 2000 The nature conservation value of European traditional farming systems. *Environmental Reviews*, **8,** 149-171.

Brooker, R. (2011) The changing nature of Scotland's uplands - an interplay of processes and timescales. *The Changing Nature of Scotland* (eds. S.J. Marrs, S. Foster, C. Hendrie, E.C. Mackey & D.B.A. Thompson), pp. 381-396. TSO Scotland, Edinburgh.

Brown, I., Poggio, L., Gimona, A. & Castellazzi, M. (2011) Climate change, drought risk and land capability for agriculture: implications for land use in Scotland. *Regional Environmental Change*, **11** 503-518.

Buenestado, F.J., Ferreras, P., Blanco-Aguiar, J.A., Tortosa, F.S. & Villafuerte, R. (2009) Survival and causes of mortality among wild Red-legged Partridges *Alectoris rufa* in southern Spain: implications for conservation. *Ibis*, **151,** 720-730.

Bullock, J.M., Aronson, J., Newton, A.C., Pywell, R.F. & Rey-Benayas, J. M. (2011) Restoration of ecosystem services and biodiversity: conflicts and opportunities. *Trends in Ecology and Evolution*, **26**, 541-549.

Callaway, R. M. & Maron, J. L. (2006) What have exotic plant invasions taught us over the past 20 years? *Trends in Ecology and Evolution*, **21**, 369-374.

Carroll, B., Russell, P., Gurnell, J., Nettleton, P. & Sainsbury, A.W. (2009) Epidemics of squirrelpox virus disease in red squirrels (*Sciurus vulgaris*): temporal and serological findings. *Epidemiology and Infection*, **137**, 257-265.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M. & Hannon, B., (1997) The value of the world's ecosystem services and natural capital. *Nature*, **387**, 253-260.

Daniell, T.J., Husband, R., Fitter, A.H. & Young, J.P.W. (2001) Molecular diversity of arbuscular mycorrhizal fungi colonising arable crops. *FEMS Microbiology Ecology*, **36**, 203-209.

Defra (2006) *UK National Action Plan on Farm Animal Genetic Resources. Presented to Defra and the Devolved Administrations by the National Steering Committee for Farm Animal Genetic Resources.* Defra, London.

Diaz, S., Tilman, D., Fargione, J., Chapin, F. S. I., Dirzo, R., Kitzberger, T., *et al.* (2005) Biodiversity regulation of ecosystem services. *Ecosystems and human well-being: Current state and* *trends: Findings of the Condition and Trends Working Group, Washington (D.C.)* (eds R. Hassan, R. Scholes & N. Ash), pp. 297-329. Island Press, Washington D.C.

EASAC (2009) *Ecosystem services and biodiversity in Europe*. The European Academies Science Advisory Council, London.

EEA (2005) *Environmental policy integration in Europe*. *State of play and an evaluation framework.* EEA technical report no. 2/2005. European Environment Agency, Copenhagen.

Ellis, J. (2012) Lichen epiphyte diversity: A species, community and trait-based review. *Perspectives in Plant Ecology, Evolution and Systematics*, **14,** 131-152.

Ellis, C.J., Coppins, B.J., Dawson, T.P. & Seaward, M.R.D. (2007) Response of British lichens to climate change scenarios: trends and uncertainties in the projected impact for contrasting biogeographic groups. *Biological Conservation*, **140**, 217-235.

Ellstrand, N.C. & Elam, D.R. (1993) Population genetic consequences of small population size: implications for plant conservation. *Annual Review of Ecology and Systematics*, **24**, 217-242.

Fraser Darling, F. & Morton Boyd, J. (1964) *The Highlands and Islands*. Collins, London.

Gabriel, D., Thies, C. & Tscharntke, T. (2005) Local diversity of arable weeds increases with landscape complexity. *Perspectives in Plant Ecology, Evolution and Systematics*, **7**, 85–93.

Gilbert, L., Maffey, G., Ramsay, S.L. & Hester, A.J. (2012) The effect of deer management on the abundance of *Ixodes ricinus* in Scotland. *Ecological Applications*, **22**, 658-667.

Haddad, N.M., Crutsinger, G.M., Gross, K., Haarstad, J. & Tilman, D.(2011) Plant diversity and the stability of foodwebs. *Ecology Letters*, **14**, 42-46.

Hawes, C.**,** Haughton, A.J., Bohan, D.A. & Squire, G.R. (2009) Functional approaches for assessing plant and invertebrate abundance patterns in arable systems. *Basic and Applied Ecology*, **10**, 34-47.

Henle, K., Alard, A., Clitherow, J., Cobb, P., Firbank, L., Kull, T., *et al.* (2008) Identifying and managing the conflicts between agriculture and biodiversity conservation in Europe: a review. *Agriculture, Ecosystems & Environment* **124**, 60-71

Hill, M.O. & Preston, C.D. (1998) The geographical relationships of British and Irish bryophytes. *Journal of Bryology*, **20**, 127-226.

Hoffman, A. A. & Sgrò, C.M. (2011) Climate change and evolutionary adaptation. *Nature*, **470**, 479–485.

Howard, B.M., Hails, R.S., Watt, A., Potschin, M. & Haines-Young, R. (2011) *Considerations in environmental science and management for the design of natural asset checks in public policy appraisal.* Paper presented at a workshop hosted by Defra, 11th May 2011. Defra Project Code NE0122.

Hughes, A.L, & Yeager, M. (1998) Natural selection at major histocompatibility complex loci of vertebrates. *Annual Reviews Genetics*, **32**, 415-434.

Keith, A. M., Van der Wal, R., Brooker, R. W., Osler, G. H. R., Chapman, S. J. & Burslem, D. F. R. P. (2008) Increasing litter diversity reduces variability in a decomposer system. *Ecology,* **89,** 2657-2664.

Kenyon, F., Greer, A.W., Coles, G.C., Cringoli, G., Papadopoulos, E., Cabaret, J., *et al.* (2009) The role of targeted selective treatments in the development of refugia-based approaches to the control of gastrointestinal nematodes of small ruminants. *Veterinary Parasitology,* **164**, 3-11.

Landis, D., Wratten, S. & Gurr, G. (2000) Habitat management to conserve natural enemies of arthropod pests in agriculture. *Annual Review of Entomology*, **45**, 175-201.

Leberg, P.L. & Firmin, B.D. (2008) Role of inbreeding depression and purging in captive breeding and restoration programmes. *Molecular Ecology*, **17**, 334–343.

Losey, J.E. & Vaughan, M. (2006) The economic value of ecological services provided by insects. *Bioscience*, **56**, 311-323.

Lundholm, J.T. (2009 Plant species diversity and environmental heterogeneity: spatial scale and competing hypotheses. *Journal of Vegetation Science*, **20**, 377-391.

Mace, G. M., Norris, K. & Fitter, A. H. (2012) Biodiversity and ecosystem services: a multilayered relationship. *Trends in Ecology & Evolution,* **27,** 19-26.

McInnes, C.J., Wood, A.R., Thomas, K., Sainsbury, A.W., Gurnell, J., Dein, F.J., *et al.* (2006) Genomic characterization of a novel poxvirus contributing to the decline of the red squirrel (*Sciurus vulgaris*) in the UK. *Journal of General Virology,***87**, 2115–2125.

Menges, E.S. (2008) Restoration demography and genetics of plants: when is a translocation successful? *Australian Journal of Botany*, **56**, 187-196.

Mineau P. & Whiteside M. (2013) Pesticide acute toxicity is a better correlate of U.S. grassland bird declines than agricultural intensification. *PLoS ONE*, **8,** 2: e57457. doi:10.1371/journal.pone.0057457.

Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R.E., Lehner, B., *et al.* (2008) Global mapping of ecosystem services and conservation priorities. *Proceedings of the National Academy of Sciences of the United States of America*, **105**, 9495-9500.

Nelson, J.L., Zavaleta, E.S. & Chapin, F. S., III (2008) Boreal fire effects on subsistence resources in Alaska and adjacent Canada. *Ecosystems*, **11**, 156- 171.

Newton, A.C., Begg G.S. & Swanston, S. (2009) Deployment of diversity for enhanced crop function. *Annals of Applied Biology*, **154**, 309–322.

Porter, J., Costanza, R., Sandhu, H., Sigsgaard, L. & Wratten, S. (2009) The value of producing food, energy, and ecosystem services within an agro-ecosystem. *Ambio*, **38**, 186-193.

POST (2012) *POST Note 408, Seeking Sustainability.* Parliamentary Office of Science and Technology, London**.**

Post, D. (2012) The long and short of food-chain length. *Trends in Ecology and Evolution*, **17**, 269-277.

Preston, C.D., Harrower, C.A. & Hill, M.O. (2011) Distribution patterns in British and Irish liverworts and hornworts. *Journal of Bryology*, **33**, 3-16.

Pritchard, G.C., Forbes, A.B., Williams, D.J., Salimi-Bejestani, M.R, & Daniel, R.G. (2005) Emergence of fasciolosis in cattle in East Anglia. *Veterinary Record*, **157**, 578-582.

Redpath, S.M., Young, J., Evely, A., Adams, W.M., Sutherland, WJ., Whitehouse, A. *et al.* (2013) Understanding and managing conservation conflicts. *Trends in Ecology & Evolution*, **28**, 100-109.

Reed, D.H. & Frankham, R. (2003) Correlation between fitness and genetic diversity. *Conservation Biology*, **17**, 230-237.

Sawyer, H., Nielson, R.M., Lindzey, F. G., Keith, L., Powell, J.H. & Abraham, A.A. (2007) Habitat selection of Rocky Mountain elk in a nonforested environment. *Journal of Wildlife Management*, **71**, 868-874.

Seivwright, LJ., Redpath, S., Mougeot, FR. & Hudson, PJ. (2004). Faecal egg counts provide a reliable measure of *Trichostrongylus tenuis* intensities in free-living red grouse *Lagopus lagopus scoticus*. *Journal of Helminthology*, **78**, 69-76.

Smith, R.K., Jennings, N.V. & Harris, S. (2005) A quantitative analysis of the abundance and demography of European hares *Lepus europaeus* in relation to habitat type, intensity of agriculture and climate. *Mammal Review*, **35**, 1-24.

Sutherland, I.A & Leathwick, D.M (2011) Anthelmintic resistance in nematode parasites of cattle: a global issue? *Trends in Parasitology*, **27**, 176-181.

The Royal Society (2009) *Reaping the benefits: Science and the sustainable intensification of global agriculture*. The Royal Society, London

Tscharntke, T., Clough, Y., Wanger, T.C., Jackson, L., Moetze, I., Perfecto, I., *et al.* (2012) Global food security, biodiversity conservation and the future of agricultural intensification. *Biological CConservation*, **151**, 53-59

Tulp, M. & Bohlin, L. (2002) Functional versus chemical diversity: is biodiversity important for drug discovery? *Trends in Pharmacological Sciences*, **23,** 225-231.

Unsicker, S.B., Franzke, A., Specht, J., Koehler, G., Linz, J., Renker, C. *et al.* (2010) Plant species richness in montane grasslands affects the fitness of a generalist grasshopper species. *Ecology*, **91,** 1083-1091.

Van der Putten, W., Van Dijk, C. & Peters B.A.M. (1993) Plant-specific soil-borne diseases contribute to succession in foredune vegetation. *Nature*, **362**, 53 – 56.

Van Grunsven, R.H.A., Van der Putten, W.H., Bezemer, T.M., Tamis, W.L.M., Berendse, F. & Veenendaal, E.M. (2007) Reduced plant-soil feedback of plant species expanding their range as compared to natives. *Journal of Ecology*, **95**, 1050-1057.

Wang, L., Wang, D., He, Z., Liu, G. & Hodgkinson, K.C. (2010) Mechanisms linking plant species richness to foraging of a large herbivore. *Journal of Applied Ecology*, **47**, 868-875.

Wilson, A.J & Mellor, P.S (2009) Bluetongue in Europe: past, present and future. *Philosophical Transactions of the Royal Society of London B-Biological Sciences*, **364**, 2669-2681.

Winqvist, C., Bengtsson, J., Aavik, T., Berendse, F., Clement, L.W., Eggers, S. *et al.* (2011) Mixed effects of organic farming and landscape complexity on farmland biodiversity and biological control potential across Europe. *Journal of Applied Ecology*, **48**, 570-579.

Woodcock, B.A. & Pywell, R.F. (2010) Effects of vegetation structure and floristic diversity on detritivore, herbivore and predatory invertebrates within calcareous grasslands. *Biodiversity and Conservation*, **19**, 81-95.

Zhu, F., Qin, C., Tao, L., Liu, X., Shi, Z., Ma, X., *et al.* (2011) Clustered patterns of species origins of nature-derived drugs and clues for future bioprospecting. *Proceedings of the National Academy of Sciences of the United States of America,* **108,** 12943-12948.

Chapter 5: Sustainable Water Management

5.1 Summary

Prioritised ecosystem services for this Broad Policy Goal are water cycling (supporting service), water detoxification and purification (regulating service), and water supply (provisioning service). The final overarching 'good' delivered by these services is water of sufficient quality and quantity in the right location for a given end user.

We have found it difficult to separate out these services and consider independently how they are underpinned by biodiversity and biophysical processes. To deal with this close interconnectedness, this chapter deviates from the structure used in the other broad policy goal chapters (Ch. 2-4). Instead, it focuses on two key aspects of sustainable water management: water quality and quantity. Delivery of both involves elements of all three ecosystem services.

With respect to water quantity

- Climate, topography, geology and physical processes play a very substantial role in determining quantity.
- Perhaps the most critical aspect of biological processes underpinning water quantity is the occurrence of specific habitats and ecosystems rather than biodiversity *per se*.
- Within these habitats it is the role of particular organismal groups, in particular vascular plants and bryophytes, that are known to have the biggest impact on water quantity. However, other groups such as soil fungi may have substantial yet currently unquantified roles.
- Native ecosystems (natural or semi-natural habitats) tend to have a greater beneficial impact on water quantity than those comprised of or dominated by non-native organisms.

With respect to water quality

- Land management can determine the functioning of biophysical processes that regulate water quality, e.g. water penetration.
- There is a much greater relative role for biological processes in regulating water quality. As for water quantity, the physical process of water penetration (prior to detoxification) may be more dependent on the occurrence of specific ecosystem types rather than biodiversity *per se*.
- However, different pollutants must be detoxified and (if possible) dealt with by different components of ecosystems. To this extent, broad biodiversity is important in enabling a wide range of potential pollutants to be handled. However, there is far less certainty about the role of biodiversity within key detoxifying groups.
- Different habitats deliver different components of the processes within the water cycle that enhance water quality.
- Although the role of biodiversity *per se* is unclear in terms of regulating the sustainability of biochemical processes, the recent occurrence of new pollutants indicates the potential for apparently redundant components of biodiversity to be of future use in detoxification processes.

Overall

- Uplands are key areas for water capture and storage, as well as for purification. Although purification processes are also important in lowland ecosystems there is much greater dependency of lowland users on upland systems than *vice versa*.

- The level of dependency between upland and lowland systems, albeit one-way, is possibly much greater than for many other ecosystem services. The scale needed for appropriate planning for the delivery of sustainable water management might be much larger (e.g. across entire catchments) compared to the delivery of services important for other broad policy goals.

5.2 Definition of the broad policy goal

The broad policy goal of Sustainable Water Management is a major driver of much national and international legislation. Most notable is the *Water Framework Directive* (2000/60/EC) (WFD) which was introduced in 2000 and applies to all rivers, lochs, estuaries and coastal waters as well as water under the ground. This Directive has formed the backbone of all subsequent water policies both internationally and in Scotland. In Scotland the WFD was implemented by the *Water Environment and Water Services (Scotland) Act* 2003 (WEWS). Central to this legislation is the protection of available water resources, through flood and drought mitigation measures, sustainable water use and the reduction of pollution. However, the Scottish Government have pledged to embed the aims of the WFD in all relevant policy and regulatory areas.

There is a spectrum of other Scottish Government policies that focus on particular aspects of sustainable water management that are of relevance to biodiversity and biophysical processes. For example, a key aim of the *Bathing Waters (Scotland) Regulations* 2008 is to improve microbiological standards. The *Flood Risk Management Bill* 2009 identifies the need for an integrated approach across land and water management. The Scottish Government has established Action Programmes to reduce and prevent nitrate contamination of waters in Nitrate Vulnerable Zones. Other policies include the *Urban Waste Water Treatment (Scotland) Regulations* 1994 and amended *Urban Waste Water Treatment (Scotland) Amendment Regulations* 2003, *The Water Supply (Water Quality) (Scotland) Regulations* 2001, *Water Quality (Scotland) Regulations* 2010, *Shoreline Management Plans*, *Climate Change Adaptation Framework*, *River Basin Management Plans* and the recently introduced (June 2012) *Water Resources (Scotland) Bill*.

The *Land Use Strategy* (LUS) recognises that the provision of clean water is fundamental to sustainable land use and that inappropriate activities on land can profoundly impact the water environment. Water management is described in the LUS as delivery of a healthy water environment "for the benefit of both people and wildlife which depend on it and the economic activities it supports". It is important to note that the LUS applies to all inland freshwaters up to, and including, inshore coastal waters, but not the marine environment. The LUS identifies that the "capacity of land to regulate water supplies (will be) increasingly valued as the climate changes and extreme weather events become more frequent". The strategy refers back to the WFD and the continued importance of implementing this legislation.

5.3 Prioritised ecosystem services

The ecosystem services prioritized for sustainable water management are water cycling, water detoxification and purification, and water supply. In the UK NEA water cycling is considered a supporting service, and water detoxification and purification are considered as regulating services, while water supply is considered a provisioning service. Water supply may also be considered as a cultural service with the provision of religious, educational and tourism-related goods. Although the goods delivered by water supply as a cultural service are not considered in detail here, we return to briefly discuss this aspect of water in Section 5.5 (Interactions between Ecosystem Services).

The waters sector in particular has been the focus of ecosystem service valuation research, not least because of the readily-recognised value of wetland-derived 'goods'. We can assign economic values to the ecosystem services provided by water cycling to help with an appreciation of their importance. Solar equivalent joules have been converted to world emdollars (EM$; a measure of the money that circulates in an economy as the result of some flow or process) in order to estimate the economic value of various processes within the water cycle. Ecosystem services related to water include rainfall estimated as EM$ 0.13/m^3, evapotranspiration as EM$ 0.21/m^3, surface runoff as EM$ 0.42/m^3, groundwater flow as EM$ 3.64/m^3 and aquifer recharge as EM$ 4.32/m^3 (Watanabe & Ortega 2011). Exploitable water resources in Scotland are equivalent to 16,000 m^3 yr^{-1} per person, and therefore it could be argued that the combined value of the water cycle for each person in Scotland per year can be up to £89,000 (Gilvear *et al.* 2002; Watanabe & Ortega 2011), although this figure depends on the possibility of realising the value of the relative over-supply.

Any changes in water supply will have very wide reaching impacts on the delivery of many 'goods' and services. Water supply *in situ* provides hydropower, recreation, transportation, fish and other freshwater products, aquaculture and the provision of water for estuaries and other aquatic habitats. Recreational uses include fishing, water skiing, canoeing/kayaking, swimming and sailing. Freshwater fishes are of considerable importance to UK ecosystem services and society as a whole. Wild UK freshwater fisheries have a significant economic value. Scotland contributes 80 % of the UK's total production of freshwater fish and shellfish including Atlantic salmon (*Salmo salar*) in coastal waters and rainbow trout (*Oncorhynchus mykiss*) in freshwaters. Recreational fishing, however, is the most important economic consideration for UK freshwater fisheries. Coarse fisheries alone contribute an estimated £850 million *per annum*, with an additional estimated £132 million in the market value of Salmonid fishing rights alone (UK NEA). For Scotland, recreational fishing has been estimated to generate around £150 million per annum (UK NEA). The growing aquaculture of Atlantic salmon and rainbow trout are further significant economic enterprises in the UK. In Scotland farmed salmon in 2011 was worth around £584.7 million to the economy (Scottish Government 2012). Diverted water supply provides water for municipal, agricultural, commercial, industrial, and thermoelectric power generation uses. In order to meet the needs of a growing and wealthier population, the demand for potable water is likely to increase in the future.

Understanding how the final 'good' of supplied water is underpinned by biophysical processes and biodiversity is critical during the current period of rapid environmental and socio-economic change. Inaccurate assessment of water supply needs has led to deficiency in supply during times of natural water shortage, for example during prolonged periods of drought. It is worth noting that the UK is not strictly self-sufficient in water. Approximately 62% of the UK's net annual water demand of 102 billion cubic metres is met by overseas sources through embedded (virtual) water, 75% of which is used for the production of agricultural biomass (Chapagain & Orr 2008). The demand for increasing food security – which may include increased agricultural production within the UK – may add further upward pressure to the demand for water. Over the last forty years rainfall in Scotland has increased substantially, especially during the winter months in the western Highlands (Barnett *et al.* 2006). Seasonality and intensity have also shifted, with the occurrence of wet winters (Nov-Apr) becoming more likely, as well as an increase in the intensity of rainfall. These changes to the water cycle have altered Scotland's hydrological regime such including the storage and flux of water through changes in river flow patterns, leading to increased flood and drought risk. As a result the capacity for various habitats to provide any ecosystem services influenced by the water cycle may have changed.

The interrelationships between the three prioritised ecosystem services for this broad policy goal are complex. The water cycle links planetary components of water, land and atmosphere through a number of complex processes such as precipitation, runoff, infiltration and evaporation (UNEP 2009). Solar energy, crustal heat, and evapotranspiration from terrestrial and aquatic ecosystems leads to the formation of clouds and precipitation which leads to surface runoff feeding aquatic ecosystems, or infiltration through the soil, generating groundwater flow. Groundwater flow is important to feed river basins during dry seasons and recharge aquifers for long-term water storage. Water detoxification and purification are elements of the water cycle, and the whole water cycle in turn supports the provisioning service of water supply. Water moves through and is altered by any ecosystem; this movement affects both the quality and quantity of service delivery dependent upon its location.

The processes involved in delivering the prioritised ecosystem services relevant to the Broad Policy Goal of sustainable water management are therefore clearly highly interconnected, arguably to a much greater extent than for any of the other Broad Policy Goals considered in this review. Consequently it is hard to separate out and consider *independently* how the delivery of the prioritised ecosystem services is influenced and/or underpinned by biodiversity and biophysical processes. We must therefore consider how this underpinning operates "in the round" and with respect to the final 'good' of supplied water (rather than trying to identify the 'goods' delivered by each service independently). However, we can disaggregate this assessment to some extent, as it is two central components of water supply - quality and quantity - that are vital for delivery of the final general 'good' of supplied water (i.e. clean water in sufficient quantities in the right location). In this chapter, therefore, we focus on understanding how biodiversity and biophysical processes underpin and help to deliver both water quality and quantity.

5.3.1 Water quantity

Scottish lochs store almost 35 billion cubic metres of water, while a further 42 billion is stored in soils (UKNEA). Scotland's contribution to the UK's water supply is disproportionately high due to topography and climate, although this contribution remains only "potential" until the infrastructure is available to enable export. Seventy per cent of the area of freshwater habitat in the UK resides in Scotland; this is 90% by volume (UKNEA). Furthermore, the uplands provide 70% of that water supply (Environment Agency 2012) while consumption is highest in the lowlands. A concern is that there have been changes in the hydrological regimes of Scottish rivers that reflect variations in rainfall patterns associated with changes in climate since the 1960s (UKNEA). River flow is becoming more seasonal, with increasing discharge in winter months (SEPA 2012). Scotland has not suffered the same declines in demand for water that have been experienced at a national scale and attributed to declining industrial requirements and privatisation which has largely improved leakage. Public water abstractions currently supply 2.4 million cubic metres of water daily, only 4.5% less than in 2002-3. Leakage in Scotland still accounts for around 38% of diverted water supplies, and possibly up to as much as 50% (Scottish Water 2003, 2012).

5.3.2 Water quality

Water quality is a measure of the chemicals, pathogens, nutrients, salts and sediments in surface and groundwater. The importance of water detoxification and purification to drinking water is obvious, but quality is an important attribute of all other hydrologic services as well.

An overall aim for Scotland is for 97% of water bodies to reach 'good' or 'high' ecological status by 2027. Based on SEPA's water quality data for 2010, 63% of Scotland's water bodies achieve this status, with groundwaters attaining moderately better status than surface waters (SEPA 2010). The major causes of pollution in Scottish rivers are sewage effluent, diffuse agricultural pollution, acidification, urban drainage, mine drainage and point source agricultural pollution. These drivers of low water quality are more substantial in lowland than in upland systems. In more densely-populated areas, particularly around the central belt and on the east coast, sewage effluent and urban drainage are the main sources of pollution. Mine drainage affects former coal mining areas within southern and central Scotland. Within rural areas, particularly those with more intensive agriculture, diffuse and point source pollution are most prevalent (Gilvear *et al*. 2002).

Water pollutants come in a variety of forms: chemicals, pathogens, nutrients, salts and sediments. Prior to 1990, water bodies had been particularly susceptible to elevated nutrient loading, often from diffuse agricultural pollutants. Agrochemicals such as fertilisers, herbicides, pesticides and insecticides are to blame for the failure of water to reach acceptable standards. Nitrogen pollution from fertilisers increased between 1957 and 1980s but has since significantly declined. An additional problem throughout Scotland is acidification caused by emissions of sulphur and nitrogen. These emissions derive from industry and motor vehicles and are a particular problem over much of the country because our base-poor (i.e. often acidic) geology and soils are unable to neutralise acid deposition (UK NEA).

Between 2002 and 2010 there was a 72 % reduction in pesticide usage in Scotland to 1392 tonnes (SASA 2010). However novel pollutants such as endocrine-disrupting compounds, nanoparticles, personal care products, pharmaceuticals, and the effects of synthetic biology are an emerging concern. These chemicals can have wide ranging effects on many trophic levels within the food chain from invertebrates to mammals, including effects on fertility, behaviour and survival (Allner *et al*. 2010; Soin & Smagghe 2007; Oberdorster & Cheek 2001). Other pollutants including bacterial, protozoan and viral pathogens are transmitted in water; a well-documented example being *Escherichia coli* which is acquired through drinking or swimming in affected water (Quilliam *et al*. 2011). As well as chemical or organismal pollutants, excess sediment in water has a deleterious effect on quality, impacting on macro-invertebrates and higher up the food chain in fish. Sediment reduces habitat diversity for macro-invertebrates (Feld *et al.* 2011) and affects the spawning, egg and young survival of fish (Louhi *et al*., 2011).

Overall freshwater quality, and consequently the ecological status of our water bodies, has improved over recent decades and particularly within the last 10-15 years. This is the combined result of legislative controls to tackle point source pollution through limiting industrial pollution and improving domestic sanitation, as well as reducing diffuse pollution from agricultural fertilisers. Since the 1980s, water quality has improved in the uplands because lower atmospheric pollution levels in these areas enable terrestrial ecosystems to buffer lakes and streams against acidification and nitrate leaching. In the lowlands, water quality improvements have largely been driven by better control of point source pollution rather than improved ecosystem regulation of diffuse pollutants. Recent reductions in the intensity of land management for agriculture have also contributed. There have been economic benefits in terms of reducing the cost of potable water, as well as having direct health benefits. However, improvements in freshwater and riparian habitats are likely to be longer term (up to 40 years), and there are still locally problematic areas in estuaries and coastal waters. Furthermore, there has been a recent increase in dissolved organic carbon in rivers, which degrades water quality (Holden *et al.* 2012).

The final 'good' delivered by the three prioritised ecosystem services – water of an adequate quality and volume in the right location - does not differ between the Scottish uplands and lowlands, although the level of demand and end use clearly does. But perhaps more so than for any other service there is a directional flow of the 'good' from uplands to lowlands and consequently a strong one-way dependency between upland and lowland systems. Supply is generally high in the uplands (because of topography) and the quality of the water used in the uplands is mainly dependent on processes within upland systems (Gilvear *et al*. 2002). In contrast in the lowlands demand is often greater, but dependent to a large part on water capture, storage, and transport processes in the uplands. Furthermore the quality of water delivered in the lowlands is the summation of actions occurring in both upland and lowland areas (pollution events vs. purification processes).

5.4 How do biodiversity and biotic and biophysical processes underpin these services and 'goods'?

The water cycle across Scotland is strongly influenced by topography, geographic variation in rainfall, the distribution of vegetation types, and the underlying soils and parent materials.

5.4.1 Water quantity

Climate, soils, topography, geology and vegetation processes play key roles in regulating water quantity, although the role of biodiversity (particularly genetic or species diversity) appears form the evidence available to be relatively limited. At a basic level topography and geology sets the framework in terms of water movement (for example the location of key areas of rainfall, the location of lochs and rivers etc.) on which ecosystems then act. Rock type influences not only quantity through permeation, but also quality through influencing chemistry and the amount of fine sediment (Cresser *et al.* 2000; Bilotta *et al.* 2012). In addition, topography and geology set limits on biological processes which then contribute to determining the types of habitats and ecosystems capable of developing (as discussed in previous chapters, for example with respect to crops and wild species diversity). The scale of the influence of the biological components of the system relative to the geological or topographic components is considerable: for example Scotland's soils, in particular peats, store an estimated 42 billion m^3 of water, which is more than the combined total volume of Scotland's lochs.

The most critical aspect of natural systems for the regulation of water quantity may be the presence and arrangement of particular ecosystems or habitats, rather than either the overall diversity of habitats or of the organisms within them. Certain habitats have specific functions within or impacts on the water cycle. Most ecosystems reduce water quantity because vegetation consumes water through transpiration: plants effectively trade water for biomass (open stomata on a plants leaf surface allow the influx of carbon dioxide, which is then captured via photosynthesis, but at the same time release water vapour to the atmosphere). However, not all ecosystems and the vegetation types contained therein consume water at the same rate. Young vascular plants generally have disproportionately large negative impacts on water quantity, because vigorously growing vegetation tends to use more water per unit biomass than mature vegetation. Trees and woody vegetation generally use more water overall than shorter vegetation because of their greater height and rooting depth; there can be as much as a 45% reduction in stream flow when grasslands are converted to forest (UK NEA).

But trees can also influence the microclimate so as to enhance water quantity. For example, canopy shading can reduce bare soil evaporation, and deforestation removes the ability for forests to intercept rainfall and release it gradually through groundwater discharge, with the maintenance of river flows during dry periods being significantly affected (Eftec 2005). Some other ecosystems also have positive effects on the regulation of water quantity: wetlands slow down the movement of water by temporarily retaining it and therefore can provide an important service for downstream flood regulation (UK NEA). In addition by releasing water over a longer period of time they are also important for water provision during low flow periods, and for the recharging of groundwater aquifers (Eftec 2005). Water supply is strongly modulated by vegetation surface properties such as through-flow and stem-flow. Fog or rain that is intercepted by a vegetated canopy can drip to the ground or evaporate directly from leaf surfaces. Vegetation also intercepts snowfall, and sublimation from the canopy can reduce average snow accumulations by up to 15% in forested catchments compared to open catchments (Smith & Lyle 1979). Therefore, the vegetation and ecosystem types within a watershed, together with their location and age, are clearly important in regulating water supply, but determining the optimum combination of these elements for maximising water quantity is a complex challenge, and may be related to prevailing climatic conditions.

In general native ecosystems provide greater water supply benefits than replacement ecosystems characterised by alien species. Invasive plant species are often fast growing and tend to have higher evapo-transpiration rates, thus reducing groundwater supplies (Richards *et al.* 2008). Impacts of vegetation may also be scale dependent.

Finally, organismal groups other than vascular plants can have important impacts on water quantity. For example, burrowing and casting activities by soil biota contribute to the creation of surface roughness and porosity, regulating the retention of water from runoff in the soil profile (Langmaack *et al.* 2001; Leonard & Rajot 2001; Lavelle *et al.* 2006 in Harrison *et al.* 2010). The mycelium of soil fungi is responsible for soil water absorption and retention capacity. However, overall effects on water quantity from fungi are poorly understood. In contrast the influence of bryophytes is well known: they are often drought-tolerant (poikilohydric) and have very high water retention properties – up to 1,500% of their dry weight. *Sphagnum* species in particular are known for their water-retention capabilities (and the water-logged and anoxic conditions associated with *Sphagnum* bogs, for example, are key locations for long-term C-storage, as discussed in Ch. 2). They are therefore of significance in areas of high rainfall, as this can be absorbed by the plants (if not already saturated) thereby preventing rapid runoff and flooding, and maintaining humidity during dry seasons (Jones *et al.* 2002).

All aspects of the biophysical processes underpinning water quantity differ between the uplands and lowlands. Upland areas receive more rainfall because of the effect of mountain ranges (Malby *et al.* 2007), and differences in slope angle mean that water drains from upland systems more rapidly. In addition the hydrological properties of upland and lowland vegetation types differ markedly. Mountain ecosystems such as moorlands, heaths and peatlands store water in snowpack, soil, vegetation and in groundwater. However, their water retention properties may not always have a "smoothing" influence on water quantity. Peaty soils have low hydraulic conductivity: water cannot flow easily through them. During dry conditions this can limit the maintenance of base flows (Holden & Burt 2003a) whilst during heavy rainfall events runoff from blanket bog produces a 'flashy' hydrological regime because of saturation excess in overland flow and near-surface through flow (Evans *et al.* 1999; Holden & Burt 2003a, 2003b). Because water runs off rapidly and soil storage is limited, stream and river flow in upland areas are closely linked to rainfall and thaw events (Baggeley *et al*. 2009) and as a result upland habitats can be poor regulators of water supply. Artificial reservoirs are therefore needed to ensure a continuous supply of drinking water in the uplands.

5.4.2 Water quality

Physical processes such as soil infiltration are critical for water detoxification and purification, whilst organisms can play a role both in regulating these physical processes and in removing contaminants from water. Vegetation, microorganisms, and soils remove pollutants from overland water flow and from groundwater by physically trapping water and sediments, by adhering to contaminants, by reducing water speed to enhance infiltration, by stabilizing eroding banks, and by diluting contaminated water.

Watershed protection plans are premised on the ability of certain kinds of land cover to either improve water quality through enhancing filtration, or to maintain water quality through minimising addition of contaminants to the water stream. Vegetation can enhance these processes depending on its particular physical characteristics. Water moving unimpeded through a watershed presents no opportunity for environmental infiltration, whereas ecosystems with characteristics that prevent gully formation through excessive erosion, and allow naturally meandering flow, are more likely to improve water quality. Root systems stabilize soils, and vegetation cover affects the force and size of raindrops hitting the ground. When this vegetation is removed, for example by logging or applying herbicide, bare soils are exposed to surface raindrop splash, runoff, and wind, which can increase erosion substantially. If erosion occurs, nutrients and other impurities built up in ecosystems through decomposition, fertilizer application, or atmospheric deposition can then become available for entrainment in water above and below ground.

Any heterogeneous strip of vegetation that forms a barrier to sediments or removes contaminants from the water stream can be considered a buffer, regardless of its location within a watershed. The widespread establishment of riparian buffer strips has been one remedial measure implemented on the back of the WFD and associated local legislation. At least 2 m wide buffer strips are now mandatory in Scotland, although this is not always currently adhered to. As well as limiting erosion and the influx of contaminants to water courses, buffer strips can help maintain the natural hydromorphology of streams and control in-stream temperatures that enhance the processing of pollutants. Reviews show substantial variation in the effectiveness and longevity of buffers, especially at a landscape scale, but it is likely that these vegetated strips can reduce local nitrate concentrations from agricultural runoff by 5% to 30% per metre width of buffer. Buffer ecosystems can thus potentially reduce water treatment costs for downstream users (Tyler *et al.* 2012).

As for the regulation of water quantity (discussed above), the impact of different habitat or vegetation types on water quality is highly variable. Peatlands filter chemical and particulate deposits and therefore act to improve the quality of water percolating through the peat matrix into streams and rivers. Wetlands play an important role in purifying water by 'locking up' pollutants in their sediments, soils and vegetation (Revitt *et al.* 2008). In particular, high levels of nutrients such as phosphorous and nitrogen, commonly associated with agricultural runoff and sewage effluent, can be significantly reduced by wetlands. This may prevent those same nutrients from reaching unsafe levels in groundwater used for drinking purposes, as well as reducing the risk of eutrophication in surface-water bodies further downstream. For wetlands to be effective they need to cover at least 2% of the area of a watershed, so the amount as well as identity of habitats is important. Under lower intensity management, semi-natural grasslands can also be critical in the maintenance of water quality in rivers draining agricultural catchments. The soils of semi-natural grasslands are able to store significant amounts of deposited nitrogen and can also reduce the pollution of groundwater (Phoenix *et al.* 2008). Forests and other mature ecosystems generally also improve water quality in water bodies. Conversely heavily-managed ecosystems such as agro-ecosystems can have a negative impact on water quality, creating dis-benefits through nutrient pollution (Harrison *et al.* 2010; Jarvie *et al.* 2008, 2010). Both the diversity of habitat types and their relative spatial arrangement within a catchment can influence infiltration and erosion. There are complex interactions between different habitats within a landscape mosaic. Composition and spatial arrangement of these habitat patches determine their degree of connectivity, which in turn controls the flux of matter and energy among adjacent patches (Langhans *et al.* 2006).

Rather than species-level biodiversity *per se* being critical, some organismal groups such as benthic invertebrates play a substantial role in influencing sediment mobility and stability (Lecerf & Richardson 2011; Gibbons *et al.* 2010; Cardinale *et al.* 2004). Beyond the physical impacts of organisms on water, sediment and pollutant movement, there are also some key biological processes that contribute toward water purification. These include biochemical transformation of nutrients and contaminants, or the absorption of water, nutrients or pollutants from the root zones of plants. Phyto- and bioremediation efforts take advantage of the uptake and transformation of contaminants by certain plant roots and the microbial communities they support. The macrophytes and microbes that promote denitrification and other biochemical processes that improve water quality are particularly abundant in wetlands, which are so reliable at removing suspended solids, phosphorus, and nitrogen from wastewater that they are regularly integrated into treatment plans.

A wide range of different biological processes are involved in the purification of water in natural systems. The UK NEA lists the following taxonomic groups as being important for water detoxification and purification: microorganisms, non-lichen fungi, macroalgae, land plants, terrestrial invertebrates, and amphibians. As the detoxification of any given chemical or pollutant may be dependent on a particular organismal group, no single group can be considered as underpinning all detoxification processes. Consequently the relationship between biodiversity and service delivery might be overall positive but resulting from a "sampling effect", with higher overall richness of species or functional groups leading to a greater likelihood of the relevant beneficial group being present. Microbes constitute a major portion of the biodiversity and biomass in waters.

Microorganisms can purify water by biodegrading pollutants as well as by stabilising sediments (Gerbersdorf *et al.* 2008), and changes in microbial diversity can result in differences in removal efficiencies of agricultural non-point source pollutants in riparian buffer zones (Yao *et al.* 2010). Soils also direct biodegradation of organic compounds through bacterial activities (Sipos *et al.* 2005 in Harrison et al. 2010).

Differences in the impacts of biological and biophysical processes on water quality in the uplands and lowlands are driven very substantially by the different types of habitats found within these systems. These combined roles of biological processes – regulating physical water flow enabling water purification, and then subsequent biochemical processes to decontaminate water – differ between habitats and ecosystems. Upland habitats overall have better water quality characteristics as a result of limited human impacts, low weathering rates, extensive peat cover and widespread overland flow. Indeed mountain habitats such as moorlands and heaths act as a buffer against atmospheric, diffuse and point source pollutants, and thus are critical for water quality in downstream rivers and water bodies. In addition water draining from upland habitats acts to dilute downstream pollutants, thus improving downstream water quality and greatly reducing the cost of providing water suitable for human consumption. However the ability to provide this regulatory service can be compromised through disturbance and erosion, with increases in dissolve organic carbon (DOC) concentrations in water draining upland habitats adding to water treatment costs.

5.5 Interactions between Ecosystem Services

Many other ecosystem 'goods' and services are dependent on the supply of sufficient water of an appropriate quality (Table 5.1). In addition, because the provision of adequate clean water is regulated by processes operating across entire catchments, and even beyond catchments (for example in the case of diffuse pollution), the delivery of a sustainable water supply interacts with many other ecosystem services.

The multiple processes governing the delivery of water quantity and quality can interact with and sometimes conflict with one another. For example, a forest might increase water detoxification and purification whilst decreasing total water supply. The optimum combination of habitats within a catchment is dependent not only on their cover, but also their spatial arrangement and the prevailing climatic conditions. In terms of interactions with the delivery of other services and 'goods', irrespective of their use of water directly any ecosystem service which impacts on vegetation cover and the relative distribution of different habitat types will influence water supply.

For example, intensification of land management for enhanced crop or livestock production can frequently impact on the role of biological and biophysical processes in regulating and detoxifying water supplies. Intensive grazing and soil compaction in semi-natural grasslands can decrease the capacity for infiltration of water, leading to increased runoff. As a result this increases downstream flood risk and reduces the ability of these habitats to recharge the aquifers needed for long-term water storage (Weatherhead & Howden 2009). Since the 1946 Hill Farming Act, the excavation of grips (field drains) in upland habitats across the Scotland – in order to enhance their agricultural production – has led to the drainage of 1.8 Mha of moorland (Mackay *et al.* 1973; Green 1974; Condliffe 2009). The installation of field drains has increased drainage density, and has exacerbated the 'flashiness' characteristics of upland runoff (Robinson 1990). As a result the drainage of upland habitats has often been associated with downstream flood events (Leannounth 1950; Stewart 1963; Oldfield 1983). However it is difficult to establish causative relationships as the hydrological processes involved are complex (Holden *et al.* 2004; 2006).

Interactions also exist between the demands of the different stakeholder groups for water supply (some examples being given in Table 5.1). The delivery of water to these different groups can sometimes be in conflict, as they often involve different ultimate *forms* of water delivery. For example, fishermen may want riffles within a water course, whereas a rafter wants white water. Hydro schemes demand adequate reserves of water but in periods of low rainfall this can impede stream flow and have negative impacts on species of high conservation concern such as freshwater pearl mussels (Addy *et al.* 2012).

Table 5.1 *Summary of services and 'goods' provided by sustainable water management (after Potschin & Haines-Young 2011).*

Service Category	**Class**	**'Goods'**
Provisioning	Nutrition	Freshwater plant and animal foodstuffs Potable water
	Energy	Renewable abiotic energy sources
Regulating	Flow of wastes	Bioremediation Dilution and sequestration
	Flow regulation	Water flow regulation
	Regulation of physical environment	Water quality regulation
	Regulation of biotic environment	Lifecycle maintenance and habitat protection Gene pool protection
Cultural	Symbolic	Aesthetic, Heritage Religious and spiritual
	Intellectual and Experiential	Recreation and community activities

This latter case is just one example where human use can have negative impacts on the ecological status of aquatic systems: freshwater environments are subject to degradation through for example over abstraction, diffuse and point source pollution, changes in channel morphology, temperature changes due to industrial activity. This is a global problem: the top four groups of organisms facing extinction are aquatic species (UNEP, 2009). Net abstraction of water from rivers and lakes has increased over the past 50 years, although a slightly reversed trend has been obvious since 1990 (EEA 2005). This is mainly attributed to urban and agricultural expansion through increases in consumptive (e.g. food, cooking and sanitation) and non-consumptive (e.g. irrigation, hydropower and cooling) use of freshwater resources. Intensified land use often causes physical degradation of streams, rivers and lakes due to drainage, discharge regulation and flood protection. Eutrophication originating from non-point sources (nitrate), due to fertiliser and manure use in agriculture, has increased since the 1960s and has remained at a high and stable level since the early 1990s. In addition demand for natural hazard and water regulation - specifically flood protection - has increased, particularly in the large cities. The combination of all these adverse effects - water abstraction, physical modification of river courses, drainage and devastation of floodplains, and eutrophication - continues to degrade purification processes of river and floodplain ecosystems. Therefore their capacity to provide 'goods' and services is vastly reduced. The decline in quality of freshwater and aquatic ecosystems has knock-on effects for a wide range of services beyond water supply (Appendix 3).

The wide web of interactions and strong dependency of other services on water supply or aquatic habitats can have beneficial, synergistic effects for water provision. For example recreational fisheries can ensure the endurance and protection of extensive freshwater habitats and their associated wildlife. These strong interdependencies can also be beneficial in terms of developing monitoring regimes for service delivery. The concept of good ecological status is based on the use of those elements of biodiversity that benefit from clean water as an indicator of service provision. Fish species and communities are arguably the best indicators of the well-being of aquatic ecosystems, in terms of both water quality and the physical environment. Negative impacts on water quality and recent signs of improvement are indicated by biodiversity responses. The abundance of commercially important estuary and freshwater fish has significantly declined since 1950s with salmon and migratory trout showing the most marked decline. The number of young European eels (*Anguilla anguilla*) returning to rivers has fallen to 1% of historical levels since the 1980s. A major cause of these declines is the lack of invertebrate prey. Invertebrates provide food for most commercial fish species and invertebrate diversity is slow to respond to habitat improvements, but in headwaters at least it has been increasing since the 1990s (Dunbar *et al.* 2010).

5.6 Knowledge gaps

Knowledge gaps can be categorised relative to the two predominant "scales" of natural systems that appear to be important for regulating water quality and quantity.

At the scale of organismal groups, the influence of some organisms on regulatory processes is not well understood but is potentially important, for example the possible role of soil fungi in regulating water quantity. Above and beyond this, though, is the question of redundancy *versus* adaptive capacity represented by organismal diversity in general. This is particularly the case with respect to detoxification processes: are there generalist species within particular organismal groups which can cope with a wide range of pollutants, or does each pollutant need a dedicated detoxifier species or lineage? In the latter case high biodiversity (genetic and organismal) represents a safety net against the future impact of novel pollutants.

At the scale of habitats, we know that particular types of habitats can play important roles in regulating water quality and quantity, but the challenge is perhaps to understand their optimum spatial arrangement within landscapes and catchments in order to maximise service delivery. We lack precise knowledge concerning the importance of connectivity between water and the habitats it flows through, and the question of scaling also needs to be addressed since our knowledge is often developed in relatively small catchments.

Associated with this is the challenge of developing management techniques that can be applied across entire catchments, as there is clearly such a strong level of dependency of service delivery in lowlands on processes in upland systems. This challenge of optimising service delivery across potentially massive land areas is probably greatest for this broad policy goal. Notably, the management of ecosystem services related to water supply tends to be localised, whereas the beneficiaries are usually more widely distributed. For example, much of the regulations of water quality are targeted towards individual locations within upland ecosystems, while the beneficiaries are downstream of these ecosystems. This disconnect between the providers and the beneficiaries of this service requires regulatory and/or incentive schemes for land and water management, and perhaps also the application of integrated planning over a much wider spatial scale than might be the case for the provision of some other services.

References

Addy, S., Cooksley. S.L. & Sime, I. (2012) Impacts of flow regulation on freshwater pearl mussel (*Margaritifera margaritzfera*) habitat in a Scottish montane river. *Science of the Total Environment*, **432**, 318-328.

Allner, B., von der Gonna, S., Griebeler, E.M., Nikutowski, N., Weltin, A. & Stahlschmidt-Allner, P. (2010) Reproductive functions of wild fish as bioindicators of reproductive toxicants in the aquatic environment. *Environmental Science & Pollution Research*, **17**, 505-518.

Anderson, T.A., Guthrie, E.A. & Walton, B.T. (1993) Bioremediation in the rhizosphere. *Environmental Scence & Technology*, **27**, 2630-2636.

Baggaley, N.J., Langan, S.J., Futter, M.N., Potts, J.M. & Dunn, S.M. (2009) Long-term trends in hydro-climatology of a major Scottish mountain river. *Science of the Total Environment*, **407**, 4633–4641.

Barnett, C., Hossell, J., Perry, C., Procter, C. & Hughes, G. (2006) *A handbook of climate trends across Scotland.* SNIFFER project CC03, Scotland & Northern Ireland Forum for Environmental Research, 62pp.

Bilotta, G.S., Burnside, N.G., Cheek, L., Dunbar, M.J., Grove, M.K., Harrison, C., *et al.* (2012) Developing environment-specific water quality guidelines for suspended particulate matter. *Water Research*, **46**, 2324-2332.

Burkhard, B., Kroll, F., Nedkov, S. & Muller, F. (2012) Mapping ecosystem service supply, demand and budgets. *Ecological Indicators*, **21**, 17-29

Cardinale, B.J., Gelmann, E.R. & Palmer, M.A. (2004) Net spinning caddisflies as stream ecosystem engineers: the influence of *Hydropsyche* on benthic substrate stability. *Functional Ecology*, **18**, 381-387.

Chapagain, A. K. & Orr, S. (2008) *UK Water Footprint: The impact of the UK's food and fibre consumption on global water resources. Volume 1*. WWF-UK, Godalming.

Condliffe, I. (2009) Policy change in the uplands. *Drivers of Environmental Change in the Uplands* (eds A. Bonn, T. Allott, K. Hubacek & J. Stewart), pp. 59–90. Routledge, London.

Cresser, M.S., Smart, R., Billett, M.F., Soulsby, C., Neal, C., Wade, A., *et al.* (2000) Modelling water chemistry for a major Scottish river from catchment attributes. *Journal of Applied Ecology*, **37**, 171-184.

Dunbar, M., Murphy, J., Clarke, R., Baker, R., Davies, C. & Scarlett, P. (2010) *CS Technical Report No. 8/07: Headwater Streams Report from 2007*. Centre for Ecology & Hydrology, Lancaster.

EEA (2005) *Environmental policy integration in Europe. State of play and an evaluation framework.* EEA technical report no. 2/2005. European Environment Agency, Copenhagen.

Eftec (2005) *The Economic, Social and Ecological Value of Ecosystem Services: A Literature Review*. *Final report for Defra.* Eftec, London.

Environment Agency (2012) *Climate change in the uplands: safeguarding vital services.* Environment Agency, Bristol.

Evans, M.G., Burt, T.P., Holden, J. & Adamson, J.K. (1999) Runoff generation and water table fluctuations in blanket peat: evidence from UK data spanning the dry summer of 1995. *Journal of Hydrology*, **221**, 141–160.

Feld, C.K., Birk, S., Bradley, D.C., Hering, D., Kail, J., Marzin, A., *et al.* (2011.). From natural to degraded rivers and back again: A test of restoration ecology theory and practice. *Advances in Ecological Research*, **44**, 119-209.

Gerbersdorf, S.U., Jancke, T., Westrich, B. & Paterson, D.M. (2008) Microbial stabilization of riverine sediments by extracellular polymeric substances. *Geobiology*, **6**, 57-69.

Gibbons, C.N., Vericat, D. & Batalla, R.J. (2010) Relations between invertebrate drift and flow velocity in sand-bed and riffle habitats and the limits imposed by substrate stability and benthic diversity. *Journal of the North American Benthological Society*, **29**, 945-958.

Gilvear, D.J., Heal, K.V. & Stephen, A. (2002) Hydrology and the ecological quality of Scottish river ecosystems. *Science of the Total Environment*, **294**, 131-159.

Green, F.H.W. (1974) Changes in artificial drainage, fertilisers and climate in Scotland. *Journal of Environmental Management,* **2**, 107-121.

resistance Harrison, P.A., Vandewalle, M., Sykes, M.T., Berry, P.M., Bugter, R., de Bello, F., *et al.* (2010) Identifying and prioritising services in European terrestrial and freshwater ecosystems. *Biodiversity and Conservation*, **19**, 2791–2821.

Holden, J. & Burt, T.P. (2003a) Hydraulic conductivity in upland blanket peat: Measurement and variability. *Hydrological Processes*, **17**, 1227–1237.

Holden, J. & Burt, T.P. (2003b) Runoff production in blanket peat covered catchments. *Water Resources Research*, **39**, 1191.

Holden, J., Chapman, P.J. & Labadz, J.C. (2004) Artificial drainage of peatlands: hydrological and hydrochemical process and wetland restoration. *Progress in Physical Geography*, **28**, 95–123.

Holden, J., Evans, M.G., Burt, T.P. & Horton, M. (2006) Impact of land drainage on peatland hydrology. *Journal of Environmental Quality*, **35**, 1764–1778.

Holden, J., Chapman, P.J. Palmer, S.M., Kay, P. & Grayson, R. (2012) The impacts of prescribed moorland burning on water colour and dissolved organic carbon: A critical synthesis. *Journal of Environmental Management*, **101**, 92-103.

Jarvie, H.P., Haygarth, P.M., Neal, C., Butler, P., Smith, B., Naden, P.S., Joynes, A., Neal, M., Wickham, H., Armstrong, L., Harman, S., Palmer-Felgate, E.J. (2008) Stream water chemistry and quality along an upland-lowland rural land-se continuum, south west England. *Journal of Hydrology*, **350**, 215-231.

Jarvie, H.P., Withers, P.J.A., Bowes, M.J., Palmer-Felgate, E.J., Harper, D.M., Wasiak, K., Wasiak, P., Hodgkinson, R.A., Bates, A., Stoate, C., Neal, M., Wickham, H.D., Harman, S.A. & Armstrong, L.K. (2010) Streamwater phosphorus and nitrogen across a gradient in rural-agricultural land use intensity. *Agriculture Ecosystems & Environment*, **135**, 238-252.

Jones, M.L.M., Oxley, E.R.B. & Ashenden, T.W. (2002) The influence of nitrogen deposition, competition and desiccation on growth and regeneration of *Racomitrium lanuginosum* (Hedw.) Brid. *Environmental Pollution*, **120**, 371–378.

Langhans, S.D., Tiegs, S.D., Uehlinger, U. & Tockner, K. (2006). Environmental heterogeneity controls organic-matter dynamics in river-floodplain ecosystems. *Polish Journal of Ecology*, **54**, 675-680.

Langmaack, M., Schrader, S. & Helming, K. (2001) Effect of mesofaunal activity on the rehabilitation of sealed soil surfaces. *Applied Soil Ecology*, **16**, 121–130.

Lavelle, P., Decaëns, T., Aubert, M., Barot, S., Blouin, M., Bureau, F., *et al.* (2006) Soil invertebrates and ecosystem services. *European Journal of Soil Biology*, **42**, 3–15.

Leannounth, A.T.A. (1950) The floods of 12th August 1948, in south-east Scotland. *Scottish Geographical Magazine*, **66**, 147-53.

Leonard, J. & Rajot, J.L. (2001) Influence of termites on runoff and infiltration: quantification and analysis. *Geoderma*, **104**, 17–40.

Lecerf, A. & Richardson, J.S. (2011) Assessing the functional importance of large-bodied invertebrates in experimental headwater streams. *Oikos*, **120**, 950-690.

Louhi, P., Ovaska, M., Maki-Petays, A., Erkinaro, J. & Muotka, T. (2011) Does fine sediment constrain salmonid alevin development and survival? *Canadian Journal of Fisheries and Aquatic Sciences*, **68**, 1819-1826.

Mackay, C. (Chairman) and eight others (1973) *Drainage of agricultural land in Scotland.* Report of Working Group, DAFS, Edinburgh.

Malby, A.R., Whyatt, J.D., Timmis, R.J., Wilby, R.L. & Orr, H.G. (2007) Long term variations in orographic rainfall: analysis and implications for upland catchments. *Hydrological Sciences – Journal des Sciences Hydrologiques*, **52**, 276–291.

Oberdorster, E. & Cheek, A.O. (2001) Gender benders at the beach: Endocrine disruption in marine and estuarine organisms. *Environmental Toxicology and Chemistry*, **20**, 23-36.

Oldfield, P. (1983) *Drain and be damned*. The Guardian, 9th April, 1983.

Phoenix, G.K., Johnson, D., Grime, J.P. & Booth, R.E. (2008) Sustaining ecosystem services in ancient limestone grassland: importance of major component plants and community composition. *Journal of Ecology*, **96**, 894–902.

Quilliam, R.S., Williams, A.P., Avery, L.M., Maltham, S.K. & Jones, D.L. (2011) Unearthing human pathogens at the agricultural-environment interface: A review of current methods for the detection of *Escherichia coli* O157 in freshwater ecosystems. *Agriculture Ecosystems & Environment*, **140**, 354-360.

Revitt, D.M., Scholes, L. & Ellis, J.B. (2008) A pollutant removal prediction tool for stormwater derived diffuse pollution. *Water Science & Technology*, **57**, 1257-1264.

Richards, C.K., Walls, R.L., Bailey, J.P., Parameswaran, R., George, T. & Pigliucci, M. (2008) Plasticity in salt tolerance traits allows for invasive of novel habitat by Japanese knotweed s.l. (*Fallopia japonica* and *F-bohemica*, Polygonaceae). *American Journal of Botany*, **95**, 931-942.

Robinson, M. (1990) *Impact of Improved Land Drainage on River Flows*. Institute of Hydrology, Report No. 113. ISBN 0948540249. Institute of Hydrology, Wallingford.

SASA (Science and Advice for Scottish Agriculture) (2010) A comparison of pesticide usage on arable crops (includes cereal crops, oilseed rape and potatoes) over the period 1988-2010. Available at: www.**scotland**.gov.uk/SESO/Resources/ [Accessed 27.09.12].

Scottish Government (2012) *Steady growth for farmed salmon.* News release 10/09/2012. Available at: http://www.scotland.gov.uk/News/Releases/2012/09/salmonproduction10092012

Scottish Water (2003) *Annual Report and Accounts 2002/2003*. Scottish Water, Edinburgh.

Scottish Water (2012) *Annual Report and Accounts 2011/2012*. Scottish Water, Edinburgh.

SEPA (2010) *Annual report and Accounts 2009-2010*. SEPA, Edinburgh.

SEPA (2012) Trends in Scottish river water quality. [online] Available at: <www.sepa.org.uk/science_and_research/data_and_reports/water/scottish_river_water_quality.aspx> [Accessed 25.09.12].

Smith, I.R. & Lyle, A. (1979) *Distribution of freshwaters in Great Britain*. Institute of Terrestrial Ecology, Cambridge.

Soin, T. & Smagghe, G. (2007) Endocrine disruption in invertebrates. *Ecotoxicology*, **16**, 83-93.

Stewart, L (1963) *Investigations into migratory fish propagation in the area of the Lancashire River Board*. John B. Barber & Sons, Lancaster.

Tyler, H.L., Moore, M.T. & Locke, M.A. (2012) Potential for phosphate mitigation from agricultural runoff by three aquatic macrophytes. *Water Air and Soil Pollution*, **223**, 4557-4564.

UNEP (2009) *Water security and ecosystem services: The critical connection, A contribution to the United Nations World Water Assessment Programme (WWAP), Kenya.* Available (online): http://www.unep.org/themes/freshwater/pdf/the_critical_connection.pdf

Watanabe, M.D.B. & Ortega, E. (2011) Ecosystem services and biogeochemical cycles on a global scaoe: valuation of water, carbon and nitrogen processes. *Environmental Science & Policy*, **14**, 594-604.

Weatherhead, E.K. & Howden, N.J.K. (2009) The relationship between land use and surface water resources in the UK. *Land Use Policy*, **26**, S243–S250.

Yao, S.H., Liu, Y.Q., Wang, Q.H., Bo, X. & Song, D.L. (2010) Characteristics of aquatic plant roots, soil microbes and agricultural non-point source pollution mitigation in riparian buffer zones. *Chinese Journal of Eco-Agriculture,* **18**, 365-370.

Chapter 6: Discussion & Conclusions

6.1 Chapter Summary

In terms of our approach we conclude that:

- Our focus on prioritised ecosystem services, although for practical reasons, has not limited the types of services or levels of biodiversity considered.
- Consistent use of terminology is essential, as is the provision of clear definitions for key phrases and concepts.
- This review should be seen as part of a process of on-going dialogue and discussion which is helping to deliver improved and shared understanding.

In terms of the relationships between biodiversity and biological/biophysical process and the delivery of ecosystem service and 'goods', we conclude that:

- Although biological and biophysical processes clearly underpin the vast majority of ecosystem services, the role of biodiversity *per se* is unclear: in many cases it is the occurrence of particular species, functional groups or habitats that seems critical for service delivery.
- Overall, the role of biological processes is constrained by abiotic physical environment, which provides the limitations (e.g. regulation of productivity) within which biological systems operate.
- In some cases service delivery is strongly and directly regulated by the physical environment, whereas in others it is mediated by interactions between biotic and physical processes.
- The differences in the physical properties of upland and lowland systems have profound implications for the biotic processes that are possible, and hence the potential for uncoupling service delivery from any biodiversity/biophysical process underpinning.
- A simple model can be produced that may prove useful in clarifying these relationships, and understanding the impacts of different broad policy goals on the strength of biodiversity/biophysical process – ecosystem service – 'goods' relationships.
- This model suggests that, because of differences in underlying system productivity, service-based indicators are better at monitoring system health in upland rather than in lowland environments in Scotland.

In terms of knowledge gaps, key generic gaps common across broad policy goal chapters are:

- Cultural services. We do not yet have a clear understanding of how to frame cultural service concepts to explore their underpinning by biodiversity and biological/biophysical processes.
- Genetic diversity. With improved analytical techniques we are now well-placed to begin to understand the role of genetic diversity in maintaining ecosystem function and service delivery.
- Functional diversity and species redundancy. For many services it is still unclear whether it is the diversity of species or of functional groups that is important, or indeed whether it is the average functional trait value as opposed to the diversity of (variance in) trait values that is critical. Adopting an approach focussed on the concept of the community weighted mean is one potential way forward.
- Spatial arrangement of species and habitats. The issue of spatial arrangement, including the possible occurrence of scale-dependent thresholds of function, might be a commonly-overlooked aspect for many ecosystem services.
- Understanding whether the Ecosystem Approach *will* promote biodiversity conservation. The implementation of an Ecosystem Approach should be assessed both on its ecosystem service and biodiversity conservation outcomes.

6.2 An overview of the review process

6.2.1 A critique of our approach

In this final chapter we summarise our review of how biodiversity and biological and biophysical processes underpin ecosystem service delivery (and the delivery of 'goods'), discussing whether there is indeed any general predictability in these relationships. We also summarise our consideration of the interactions between ecosystem services, and the results of our contrast of upland and lowland systems. Throughout this Discussion we identify some key generic knowledge gaps. These are indicated in the text by ***bold italic blue font***. They are then summarised at the end of this Chapter.

Before summarising our findings, it is worth considering again the approach that we adopted. The aim of this review work is to help deliver the request from Scottish Government for:

> *Increased understanding of the linkages between the primary ecological and evolutionary processes, ecosystem function and ecosystem services, to inform assessment of the consequences of environmental change for the wide range of ecosystem services. (RD 1.1.2).*

By undertaking this review exercise, focussed on the underpinning of ecosystem service delivery by the natural environment (both its biotic and physical components and processes), we aimed to:

1. Improve shared understanding across the Work Packages, Themes, and Programmes about this element of the ecosystem service and Ecosystem Approach concepts.

2. Better target future research activity toward identified knowledge gaps.

The production of this review helps to achieve these aims by setting out definitions for key concepts and then further elucidating them through application to specific examples. The comparison of upland and lowland systems can also be a useful lens for achieving this improved understanding, but perhaps more importantly the upland-lowland contrast has provided what we believe to be some novel insight into generic relationships, as we discuss below (Section 6.3).

During the course of the review we have identified a large number of knowledge gaps. Some of these might have been derived at the offset from existing reviews and syntheses (e.g. from the UK NEA, and from existing published literature reviews as discussed in Ch. 1). However, we feel that we have added to these existing documents by considering these issues specifically within the context of Scottish systems, and by taking our consideration of relationships to a greater level of resolution by trying to focus on the delivery of 'goods' as well as services.

The focus on 'goods' is important: the way in which biodiversity and biological/biophysical processes underpin the delivery of a service can be very much dependent on the nature and location of the 'good'. For example, within the provisioning service of crops only insect-pollinated crops (a 'good'), which make up a relatively low volume (although high value) of Scottish crop production, are dependent on other organisms for pollination. Consequently we cannot *assume* that there is a generic relationship between biological/biophysical processes and the delivery of any one service based on the relationship that is relevant to a particular 'good'.

We have also focussed on only a subset of possible services. To be comprehensive for all 'goods' delivered by all services in upland and lowland systems would have been beyond our capacity. It is important to consider whether the prioritisation of different services would change our final conclusions. We do not think this would be the case. By taking the ecosystem services prioritised by the Ecosystem Approach Working Group, and by relating prioritisation to broad policy goals, we have addressed a range of service types and policy sectors. Our review has also covered a wide range of services and associated 'goods' in terms of the level of our current knowledge, from those where we have a reasonable level of understanding, for example in the case of soil biodiversity regulation of soil carbon (Ch. 2), to those where our understanding might be considered currently very poor, for example in the case of wild species diversity as a cultural service (Ch. 4). We have also considered relationships occurring across a range of spatial scales, from the regulation of crop production within a single field (Ch. 3) to the delivery of water across catchments (Ch. 5). The approach adopted has therefore stopped us from focussing only on those services and 'goods' where we are likely to have the best information.

It is notable that the prioritised services did not include many supporting services. This point was also raised by attendees of the Ecosystem Approach Working Group workshop. However, these services have not been overlooked by our review process. In order to work through the chain of interactions from biological/biophysical processes to the 'good' (Ch. 1), we have been forced often to consider the impact of biological/biophysical processes on a number of ecosystem processes and functions which can also be categorised as intermediate processes or supporting services. For example, we have explored how soil formation is regulated by biological processes (including biodiversity *sensu stricto*) and in turn delivers toward climate regulation through carbon storage.

A clear outcome of the review process is the importance of terminology. We have tried to apply a standard terminology throughout this document, and to provide explicit definitions and explanations of this terminology. This is essential: the field of ecosystem service research, and its uptake by a range of sectors and policy areas, is moving so rapidly that the application of terminology is constantly changing. We have tried our best to marry our terminology with the UK NEA, which acts as an important benchmark, and even if the terminology we have used differs from that of other studies, at least the clear definitions provided will help to make it clear where these differences lie.

Finally, we have produced this review as part the on-going process of dialogue and discussion that underpins the work of the Ecosystem Services Theme. We do not put it forward as a comprehensive assessment, but hope that both the specific information and the generic messages are useful to key stakeholders working on the management of the Scottish environment, or those interested in the delivery of particular broad policy goals. We are happy to discuss the review within this context, and we plan to return to the review and update it in the light of developments and improved knowledge stemming from our own research, and from that being undertaken by the wide range of research addressing these important issues both within the UK and beyond.

6.2.2 Ecosystem service interactions

The chapters focussing on the broad policy goals (Ch. 2-5) have highlighted many examples of the types of interactions associated with ecosystem services and their underpinning by biological/biophysical processes and biodiversity. These interactions occur both within and across a range of scales, and can be both synergistic and antagonistic. Table 6.1 provides examples of the different types of interaction that can occur between different components of the chain running from natural systems to the delivery of ecosystem services and 'goods' (Ch.1).

Overall, the data in the broad policy goal chapters show the considerable complexity of the relationships and interactions between the different components of the natural system-ES-'good' chain. However, handling the complexity of these interactions is at the heart of the uptake and dissemination of the Ecosystem Approach in environmental decision making. Effectively it is the lack of recognition of these interactions - of the impact of delivery of one good or service on the long-term capacity of ecosystems to deliver many other services - that can lead to the degradation of natural capital and overall service delivery. It is hoped that by describing the potential effects of plans or decisions on all relevant ecosystem services, and the economic and social consequences of these, the Ecosystem Approach (or an ecosystem approach) will integrate and align economic, social and environmental policies. The approach may also facilitate public engagement with decision-making by reflecting how places are valued, and by explaining the environmental consequences of policy or land-use decisions.

Although a seemingly impossible challenge, unpicking these interactions and the relative strengths of different relationships is essential in implementing an integrated approach to land management based around balanced delivery of ecosystem services. The application of tools such as Bayesian Belief Networks, for example, may provide approaches for starting to assess how ecosystem services can interact, and predicting how a focus on one particular service will influence the delivery of other services.

Table 6.1 *Examples of the different types of interactions that can occur within the natural system-ecosystem service-'goods' chain.*

Broad Policy Goal	Type of interaction	Mechanism
Interactions between different 'goods' delivered by a single service		
Low C economy	Negative interaction between sequestered carbon in unused fuels and production of biofuel crops	The growth of biofuel crops necessitates the use of hydrocarbon fuels, with C emissions from fuel consumption outweighing emissions benefits from biofuel production
Halting biodiversity loss	Negative interactions between red deer and Capercaillie	Management of deer may necessitate fencing, which has negative impacts on Capercaillie through bird strike.
Interactions between prioritised services within a Broad Policy Goal		
Low C economy	Negative interaction between woodland expansion and soil formation/peat development	In upland systems on organic soils plantation forestry can lead to a net loss of stored C from the system due to increased soil decomposition, despite C fixation in timber
Sustaining food production	Negative interaction between livestock and crop production, particularly in lowland systems	Land used for livestock inevitably cannot be used for crop production. This conflict does not occur in the uplands as biophysical limits restrict crop production and so promote livestock farming
Halting biodiversity loss	Negative interaction between wild species diversity as a cultural and as a provisioning service	Promotion of particular species for hunting (e.g. grouse, pheasants) has involved suppression of top predators of conservation concern (e.g. raptors)
I*nteractions between Broad Policy Goals*		
Sustaining food production & low C economy	Negative interactions between crops and soil formation	Soil cultivation practices degrade soil structure and C storage
Halting biodiversity loss & sustainable water management	Positive interaction between wild species diversity and water quality	Moorland restoration projects, for example for iconic bird species, have positive impacts on habitats that are vital for regulation of water quality (e.g. peatlands)
sustainable water management & sustaining food production	Negative interaction between water quality and crop production	Intensive crop production processes lead to soil compaction, reduce water infiltration, and enhanced sediment and nutrient loads in water courses

6.3 How do biophysical processes and biodiversity underpin ecosystem services?

This brief discussion of interactions between ecosystem services (above) demonstrates the potential complexity of managing for the delivery of ecosystem services, and of understanding the relationships in the natural systems-services-'goods' chain. However, some broad generalisations are possible. First we discuss in general terms the role of natural systems, including biodiversity *sensu stricto*, in underpinning ecosystem service delivery. We then discuss the way in which the physical environment can set restrictions – a "framework" – within which biological and biophysical processes can operate.

6.3.1 Role and redundancy of biological processes and biodiversity

As noted (Ch. 1) "Biodiversity" can refer to "nature" or it can refer to the stricter CBD definition: *"Biological diversity' means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems."* Both uses of the term "biodiversity" are common in the wider literature, but in our review we have tried to be specific about the levels of biodiversity being discussed in any given case. When not referring to biodiversity *per se*, for example when discussing simply nature, biota, or biotic/natural processes, rather than something that is specifically about *diversity*, we use these other phrases instead.

In terms of the critical level of biodiversity for delivery of a specific 'good', this is highly variable. For example, pollinator species diversity is important for production of insect-pollinated crops (Garibaldi *et al.* 2013), and habitat diversity is important for water quality. Although for many of our prioritised ecosystem services we have been able to determine the influence of some particular component of natural systems (with the exception of a few services such as intensively-farmed wind-pollinated crops), there are clearly sizable gaps in our knowledge concerning how different components of biological systems and biodiversity support different ecosystem services. It should not be assumed that because some relationships between service delivery and certain components of natural systems or biodiversity are not discussed they are unimportant: it simply means that *we do not know* whether these relationships are important.

In many cases we have a far better understanding of the response of biodiversity and biological systems to management for a service than their role in service delivery. In terms of understanding the impact of biodiversity and biological processes *on* service delivery, some identified knowledge gaps are less surprising than others. For example our ability to assess, let alone understand, the ***role of genetic diversity*** has until very recently been highly constrained by our inability to measure it. With new fast through-put DNA sequencing and increasingly cheap analytical techniques, we are now in a position to look more closely at the significance of genetic diversity for ecosystem function and service delivery. Despite being absent from the CBD definition of biodiversity, ***functional diversity*** (the diversity of functional groups) is also often highlighted as being poorly understood but likely to be of important for service delivery. Some examples of the identified role of functional diversity include soil formation, crop production, and water detoxification and purification. In these cases it is thought to be the diversity of functional groups rather than absolute species diversity which is critical to maintaining service delivery. In pursuing this research area one potentially fruitful approach may be the concept of the community weighted mean (as discussed in Ch.1), and consideration of whether it is the community's "average" functional trait values which are important for regulating ecosystem function and service delivery, or instead whether it is the diversity (variance) in functionality present that is important.

In some cases increasing biodiversity (at whatever level) promotes the delivery of goods and services, so can we say that the relationship between biodiversity and the delivery of services and goods is generally positive? This seems like a reasonable conclusion but with occasional exceptions, some of which may be specific to the Scottish environment. First, in some specific circumstances this relationship can be negative: for example increased habitat diversity or species diversity may be associated with increased pests or diseases. Similarly an influx of non-native species enhances diversity but can degrade many services, including for example water quantity and quality or wild species diversity. In addition in some circumstances high diversity (native or non-native) is explicitly not what is needed for service delivery (Mace *et al.* 2012), for example in the case of wild species diversity as a cultural service in mountain regions. The second exception to the "generally positive relationship" rule is where there appears to now be no clear relationship, and where interventions have weakened the dependency. For example, in intensive lowland crop systems the relationship between biological processes and biodiversity and service provision can be very weak or non-existent, as many of the natural processes have been replaced by anthropogenically driven processes.

The effects of biological processes and biodiversity on service delivery might impact on the sustainability of services, as well as their immediate delivery. For a number of services (e.g. soil formation, or wild species diversity) we have noted that although much of the apparently high biodiversity may appear redundant (for example soil species diversity or many species' genetic diversity), it may have a longer-term role in enabling continued service delivery in the face of environmental variability. To put it another way, this apparently redundant diversity may support system resilience. However, although this statement is often made, the ***potential role of apparently redundant biodiversity*** is perhaps one of our biggest common knowledge gaps.

Another key knowledge gap is the ***role of the relative spatial arrangement of organisms***. This appears to be vital for regulating some biological and biophysical processes and service delivery, but is not often considered. We know, for example, that the spatial arrangement of habitats across a catchment can strongly regulate water quality and quantity. Similarly the specific mosaic of habitat patches, as well as their identity, is important in supporting beneficial organisms such as predatory invertebrates at the centre of integrated pest management systems, or insect pollinators. The arrangement of genetic diversity within a crop such as barley can also have substantial implications for pest control. But how, for example, does the spatial arrangement of soil organisms influence their function?

Questions about the role of relative spatial arrangement might need to be addressed ***at a scale which is relevant to the delivery of the service or 'good'***. For example, it is the arrangement of habitats within a catchment that can influence water supply, but the arrangement of soil organisms within a field or even soil column that regulates soil formation. Studies of the associated processes therefore need to be at the appropriate scale. But scale also has implications for management as well as research: integrated catchment management aimed at balancing ecosystem service delivery might have to juggle not only the abundance of different organismal groups but also their relative distributions at a range of scales to optimise ecosystem service delivery. Differences in the spatial scale at which underpinning elements of biological systems might need to be managed could strongly influence the difficulties of achieving adequate service delivery. It may be easier, for example, to persuade a single farmer to manage a particular field to enhance soil formation than to achieve the coordinated management at a catchment scale needed to maximise conservation-relevant goods from biodiversity.

6.3.2 Physical characteristics set a framework within which biological systems operate: a simple model for understanding the potential strength of relationships

Contrasting the upland and lowland systems of Scotland demonstrates clearly the role of the abiotic physical environment in regulating biological processes, and consequently in driving spatial segregation of ecosystem services.

For some services the regulatory influence of the abiotic physical environment on biotic processes is obvious and very direct. Water supply is highly dependent on upland systems because the mountainous topography promotes rainfall. This would be true irrespective of any role of biological systems. For other services, however, service delivery results from interactions between the physical environment and biota. Most critically, perhaps, upland systems have environmental conditions that result in relatively lower absolute primary productivity compared to lowland systems: they are colder, wetter, cloudier and windier. These environmental conditions suppress the rate of biotic processes (Brooker 2011), in turn regulating two additional factors that influence final service delivery.

First, limits on primary production regulate the range of habitat types that are possible for a given location. For example, productive silvicultural and crop systems are simply not viable in upland environments. In lowland systems a wider range of habitats is possible. Lowland systems can contain heath and bogs, but also highly productive crop and forestry systems, and conflicts between services might be more frequent as a result.

The second impact of the physical environment is again related to system productivity, but in this case is indirect and mediated through human intervention. In upland systems productivity is relatively low. This means that the potential return from land management (in terms of either volume of 'goods' or money) is also relatively low. Consequently these lands are less intensively managed, not only because the physical potential for moving them to a higher productivity state is limited (they cannot exceed a given absolute threshold of productivity, at least not without an improbable amount of investment), but also because there are no financial benefits to land managers from attempting to do so. In contrast in lowland systems, because of higher primary productivity the potential returns from the land are such that it is economically viable to intensively manage, even if this necessitates replacing natural processes (lost from degrading ecosystems) with artificial processes. The productivity of the environment therefore determines both the possible habitats and associated species, and the extent to which these *can be* and *are* modified by management activity. These relationships can be summarised as a simple flow diagram (Fig. 6.1).

Figure 6.1 *A simple representation of the interrelationships between the physical component (e.g. climate, rock type), the biotic component (habitats and species), and the human component of an environment and the delivery of ecosystem services. In this simple representation, the physical component can both directly and indirectly regulate service delivery. The human component is itself regulated by service delivery (for example in terms of the income generated by land), and in turn regulates the biotic component through management practices (whose extent is in turn dependent on the system's ability to produce services).*

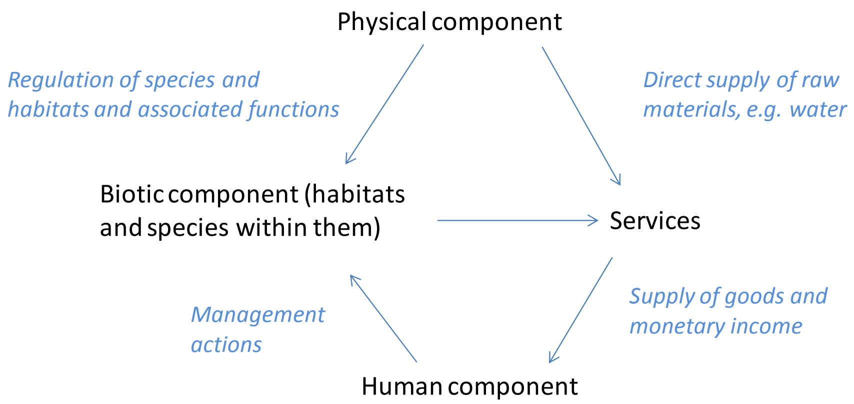
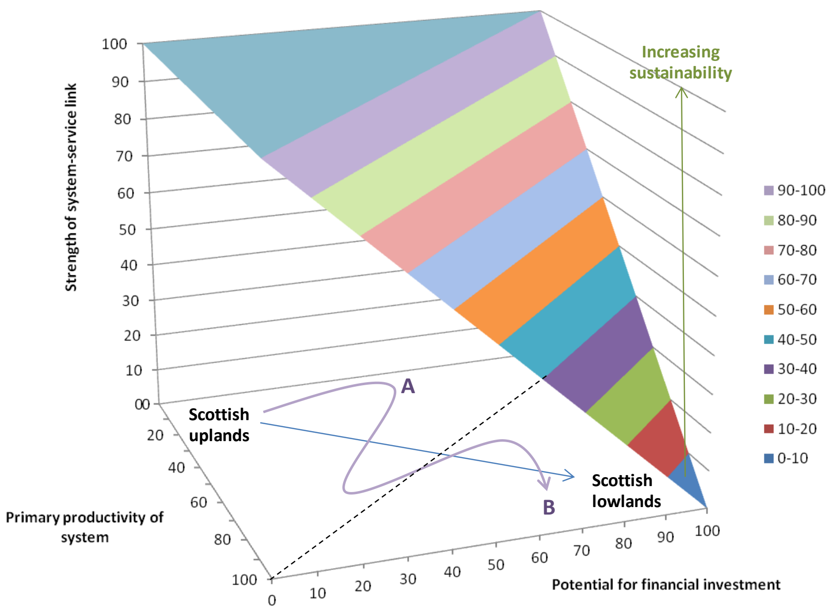


***Figure 6.2** Simple graphical model for the extent to which ecosystem service delivery <u>can be</u> detached from underlying biodiversity and biophysical processes. The two underlying drivers of this potential detachment are the primary productivity of the system, and the potential for financial investment in land management. These drivers are shown in arbitrary units from 0 (Low) to 100 (High) but are not independent. Low primary (biotic) productivity reduces income and the potential for investment in management. However, there is not always a direct positive correlation: high primary productivity does not necessarily mean high income generation – this is dependent on whether the potential income is realised. Consequently, at low primary productivity investment in management interventions is constrained, and the delivery of services is directly dependent on biodiversity and biophysical processes (there is a strong link between natural systems and service delivery). However, as primary productivity increases, potential income streams increase and consequently the strength of the link between natural systems and service delivery is reduced (shown in the response surface by a reduction in the strength of the system-service link). It is important to recognise though that the response surface represents the <u>hypothetical potential maximum decline</u> in the strength of this link: this maximum will only be realised if services are exploited and resulting money used to invest in artificial processes that replace natural ones. Note also that the model does not indicate whether the relationship will be positive or negative, simply whether in general it is likely to be strong or weak. The green arrow shows the hypothetical change needed to re-establish the link between natural systems and service delivery, and increase system sustainability. The purple arrow shows the movement of a service such as water supply across the response surface, with the final dependency on natural systems of that service being the mean value across the length of the path.*



We can extend these proposed relationships to develop a broad-brush model of the extent to which ecosystem service delivery in different environments *can be* detached from the underlying biological and biophysical processes and biodiversity (Fig. 6.2). We can work through the simple model in the context of Scottish upland and lowland systems and associated management practice.

Scottish uplands might be considered to be in the “low productivity + low financial investment” zone: for most services in these systems the link between biodiversity/biophysical processes and ecosystem service delivery is strong. For example water purification, C storage, and wild species provisioning are all strongly related to elements of the natural environment. Land management interventions, where they occur, tend to represent low investment per unit area, for example extensive livestock grazing. The productivity of even managed systems is highly dependent on “natural” biota.

Scottish lowland arable systems might represent the “high productivity + high potential for financial investment” zone. In intensive crop production systems delivery of the service has been disconnected from natural biotic and physical processes by investment in certain management activities (mechanisation and the use of agrochemicals). This investment is possible because of the high financial returns delivered by these more productive systems (and possibly also because of ease of access). Note that the model does not show whether the link between biodiversity/biophysical processes, services and goods will be positive or negative, but instead indicates the strength of the link and the potential for it to be broken or disrupted.

This model is useful in that it helps us to visualise how abiotic environmental conditions interact with biotic processes to regulate service delivery, and how in turn delivery of ‘goods’ by services can allow human intervention to uncouple service delivery from the biophysical/biological underpinning. It helps us to predict that - although varying between different types of service - in general the strength of the relationships between biodiversity and biological/biophysical processes and service delivery is likely to be stronger in upland than in lowland systems. It also demonstrates why issues of service sustainability are perhaps more critical in lowland systems, where the detachment from an underpinning by natural systems has the potential to be greatest.

So what are the lessons from this model for achieving our broad policy goals and managing for long-term ecosystem service sustainability? In general in upland habitats maintenance of the existing environment in a healthy state is essential for service delivery, as the link between system function and service delivery is tight and direct. Sustainability of service delivery can readily be equated with conservation of biodiversity. However in lowland systems, service delivery may be highly detached from the natural environment. Those cases where service delivery is highly detached represent the greatest risk for unrecognised degradation of natural capital; in such circumstances alternative functions are put in place to replace the decline in natural function, and may in turn degrade natural capital still further (for example the impact of intensive farming practice on soil structure and function). Such situations should be the focus of intensive monitoring to ensure natural capital is not being eroded, but should not be monitored simply using indices of the flow of ecosystem services (as these might be detached from the status of the underlying natural capital). In contrast, in upland systems the direct link between natural systems and the delivery of goods and services means that goods and services are likely to be good indicators of overall ecosystem health.

Much of the evidence in this review suggests that long-term sustainability of the services that have a substantial disconnect from biodiversity and biophysical processes would be promoted by increasing the strength of this link. This will provide greater long-term system resilience and reduce dependency on external inputs. A move toward increased sustainability is therefore a choice to actively strengthen the link between system function and service delivery. The evidence in this review would also suggest that this would be beneficial across multiple broad policy goals. For example, a greater reliance on integrated pest management (the use of natural predators and crop resistance to control pests and pathogens) would reduce dependency on fossil fuel inputs into agriculture. Benefits would accrue for all four broad policy goals from this shift in management, although the subtle balance of these benefits against some potential disbenefits for particular broad policy goals (e.g. reduced crop yield) would need careful exploration.

What about services whose delivery spans the gradient from unproductive uplands to highly productive lowlands, for example water supply? In this case it is possible to trace a path across the response surface of the model that mirrors the flow of the good (Fig. 6.2). The net level of connection between natural systems and goods delivered would be the mean value accrued across that path. This in turn would be dependent on the point of abstraction. If water is abstracted in the highlands, most of the delivery of the good is dependent on natural systems. If water is abstracted in the lowlands, the good might be dependent on both natural systems and on some replacement systems in the lowlands (e.g. artificial water purification). Importantly this comparison of upland and lowland systems, and the proposed generic model, is relevant certainly across temperate environments in general, where lowlands tend to be productive and uplands unproductive.

This model is an initial draft and needs further consideration and discussion. As noted throughout this review it is based on a highly-imperfect set of field data. In the model itself, further refining the investment axis would be an interesting focus for discussion between environmental scientists and economists: might a combination of return on investment plus an index of accessibility (e.g. the occurrence of transport infrastructure) be a good way of assessing the potential for service delivery to be disconnected from natural systems? For example, highly productive systems with high potential returns might not be exploited if the infrastructure does not exist to transport 'goods', or to allow access to *in situ* 'goods'. In addition, there may be some services that don't fit this model: for example the delivery of wild species diversity for nature conservation will always be highly dependent on biodiversity/biophysical underpinning. Perhaps for these services the service delivery is predictable directly from the depicted response surface? The model might also tell us something about the likely ex-system environmental footprint of service delivery. The greater the disconnect between natural systems and service delivery, the greater the need for external inputs (fossil fuels, chemicals, mechanisation) in order to maintain service provision. However, and despite being at an early stage of development, we suggest that this model might be a worthwhile focus for further discussions concerning the link between biodiversity and biological/biophysical processes and ecosystem service delivery.

6.5 Knowledge gaps

Many specific knowledge gaps have been outlined in each of the broad policy goal chapters. Some of these are relevant to particular processes in particular systems. Rather than reiterate these here, we suggest readers refer to the appropriate sections of the broad policy goal chapters. Instead, here we briefly summarise more generic knowledge gaps, some of which have already been mentioned above:

- Cultural services. At the outset of the review process we decided that we do not yet have a clear enough understanding of how to frame cultural service concepts to explore their underpinning by biodiversity and biological/biophysical processes. This represents a clear research target, especially for interdisciplinary research activities. Once we have defined more clearly the components of an environment that contribute to cultural service delivery, we can then explore how these are underpinned.
- Genetic diversity. With improved analytical techniques we are now well-placed to begin to understand the role of genetic diversity in maintaining ecosystem function and service delivery. What may be critical is the extension of techniques that are well developed in production systems, for example crop and livestock systems, to wider natural diversity. Understanding the role of genetic diversity has important implications for the biodiversity conservation sector in particular, as much current debate is focussed on the need to capture "enough" genetic diversity during conservation programmes. We might ask, how much is

enough, and is this the same for species conservation *and* the conservation of the services that depend on the species?

- Functional diversity and species redundancy. For many services it is still unclear whether it is the diversity of species or of functional groups that is important, or indeed whether it is the average functional trait value as opposed to the diversity of (variance in) trait values that is critical. There appears to be considerable redundancy within functional groups, but we need to be careful when exploring this issue to take a long enough temporal perspective. For example, only a subset of extant soil organisms might provide at any one time the functional diversity necessary to deliver soil services, but the within-group species diversity may provide long-term stability of ecosystem functions within a variable environment. Adopting an approach focussed on the concepts of the community weighted mean of traits is one potential way forward.
- Spatial arrangement of species and habitats. Although obvious for the delivery of some functions (e.g. habitat connectivity), the issue of spatial arrangement might be a commonly-overlooked aspect for many ecosystem services. The scale of spatial arrangement may be critical, as well as the possible occurrence of thresholds of habitat coverage for service delivery (due, for example, to edge effects), and the possibility that these are dependent on the processes and services being considered.

An additional key knowledge gap which has not been discussed previously concerns the overall efficacy of adopting an ecosystem approach, and the risk presented by our current shortfall in knowledge concerning the underpinning of ecosystem service delivery by biodiversity and biological/biophysical processes. Much recent nature conservation legislation is based on the assumption that conservation of natural systems will protect ecosystem service delivery. Consequent to this assumption, there is a societal benefit in conserving nature, and this in turn will help integrate – indirectly – nature conservation into many policy sectors. However, the precise extent of dependency of ecosystem service delivery on biodiversity and biological/biophysical processes is in many cases unknown. If we make substantial land management decisions based on relationships that are poorly understood, there will be potential for sectoral interests to invest effort in dismissing the evidence as incomplete and inconclusive. An analogy to this might be the considerable investment in climate-sceptic research by organisations with a desire not to have their activities curtailed by climate change legislation. This is one long-term reason why it is vital to invest effort into understanding these relationships.

A final and associated knowledge gap is whether the Ecosystem Approach will deliver against its original purpose. Its uptake and propagation have been driven by the recognition that biodiversity conservation was losing out in the policy arena (Ch. 1). By highlighting the role of biodiversity and natural systems in delivering ecosystem 'goods' and services it is hoped that the Ecosystem Approach will promote biodiversity conservation. However, we should not lose sight of the need to continue conserving biodiversity for its own sake, irrespective of the delivery of services. Consequently we should assess whether implementing an Ecosystem Approach *will* help promote biodiversity conservation, and the implementation of an Ecosystem Approach should be assessed both on its ecosystem service and biodiversity conservation outcomes.

References

Brooker, R.W. (2011) The changing nature of Scotland's uplands - an interplay of processes and timescales. *The Changing Nature of Scotland* (eds. Marrs, S., Foster, S., Hendrie, C., Mackey, E.C., Thompson, D.B.A.), pp.381-396. Scottish Natural Heritage, Edinburgh.

Garibaldi,, L.A., Steffan-Dewenter, I., Winfree, R., Aizen, M.A., Bommarco, R., Cunningham, S.A. *et al.* (2013) Wild pollinators enhance fruit set of crops regardless of honey bee abundance. *Science Published online 28 February 2013 [DOI:10.1126/science.1230200]*

Mace, G. M., Norris, K. & Fitter, A. H. (2012) Biodiversity and ecosystem services: a multilayered relationship. *Trends in Ecology & Evolution,* **27,** 24-31.

Appendix 1

The 12 Principles of the Ecosystem Approach

Taken from http://www.cbd.int/ecosystem/principles.shtml, 9th November, 2012

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 favor 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 that change is inevitable.

Ecosystems change, including species composition and population abundance. Hence, 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.

Appendix 2

Summary of notes from the Appendices to Ch. 4 of the UK NEA. The table considers the importance of different taxa for delivering recreation and conservation.

Taxonomic group	**Recreation**	**Biodiversity conservation**
Microorganisms		Pathogens as drivers of wild species' populations
Fungi – non-lichenised		Fungi can have positive or negative impacts on wild plant populations and wild fauna
Lichens	*En masse*, lichens contribute substantially to aesthetic character of celebrated GB landscapes	Lichens contribute substantially to wild species diversity; lichen-ephiphyte abundance is correlated with food source availability for birds
Phytoplankton		The foundation of virtually all marine food chains
Macroalgae		Provide feeding and nursery habitat for fish and shellfish [and presumably other coastal species]
Bryophytes	When dominating ecosystems, can be important elements of the habitat/environment, e.g. *Sphagnum* bogs or bryophyte-rich wet woodland	Bryophytes contribute significant diversity to almost all GB ecosystems; provide microhabitats for many invertebrates, which are the food for many other species (particularly in aquatic or mossy ecosystems)
Seagrass		Nursery and foraging habitat for fish, shellfish and wildfowl; act as a foundation species to enhance overall biodiversity
Land plants	Vascular plants are the framework for meaningful places, promote health and are valued in green landscapes	Wild species diversity is ultimately dependent on vascular plants
Marine and estuarine invertebrates	Marine and estuarine invertebrates matter to people, from popular scuba habitats to shells on the beach	Invertebrates form structures which provide habitats for other marine life
Terrestrial and freshwater invertebrates	Butterflies may play a role in meaningful places and socially valued land- and waterscapes	
Marine fish	Observation of marine fishes contributes to the marine diving experience	Contribute to wild species diversity
Freshwater fish	Recreational fishing is the most important economic consideration for UK freshwater fisheries	Through freshwater fisheries fish ensure the endurance and protection of extensive freshwater habitats and their associated wildlife.
Amphibians		Frogs and newts frequently feature in urban conservation and green space initiatives, and may promote conservation of other species (e.g.

		through creation of ponds)
Reptiles		Presence of reptiles helps protect habitat, particularly with respect to the rarer species.
Birds	Bird watching is a very substantial recreation activity	Birds play a major role in wild species diversity
Mammals		Primarily through their presence as a conspicuous component of our wild diversity, but secondarily through grazing and the maintenance of habitats and predation (potentially negative, e.g. mink)
Marine mammals	Main human value may be delivered through eco-tourism	As top predators, can strongly influence structure and function of communities.

Appendix 3

Assessment of the capacities of a range of water-related land cover types to provide selected ecosystem services and to support ecological integrity, adapted from Burkhard, B., Kroll, F., Nedkov, S. & Muller, F. (2012) Mapping ecosystem service supply, demand and budgets. *Ecological Indicators*, **21**, 17-29 Fig.2).