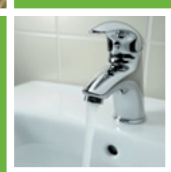
Countryside Survey



An Integrated Assessment of **Countryside Survey** data to investigate Ecosystem Services in Great Britain

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Contents

Executive Summary

Countryside Survey and 'Ecosystem Services' in Britain

- Countryside Survey (CS) provides a unique time series of data which incorporates measures of soil, water, vegetation and landscape made at the same locations in 1978, 1990, 1998 and 2007. These data provide the potential for integrated analyses of selected '*ecosystem services*', using a sampling framework which enables reporting at the GB scale.

- Increasing pressure on natural resources and declining biodiversity have led to a concern about the potential of natural and semi-natural ecosystems to provide for human requirements both currently and in the future. Termed '*ecosystem services'*, these provide a focus for policy makers seeking to ensure sustainable development.

- Ecosystem services provide a significant challenge for scientists. Difficulties arise in their definition, valuation and measurement of stocks and flows. It is problematic but of vital importance to quantify how ecosystem services interact with each other within and between ecosystems. CS data provide a unique opportunity to contribute to this challenge due to the co-location of a wide range of measurements and linked information to help interpret changes over time; some since 1978.

Approach

- The work described in this report uses data collected from the Countryside Surveys, alongside other relevant datasets to provide '*integrated assessments'* of a range of indicators[1] of ecosystem services at the national scale. Understanding ecosystems at any scale is a significant scientific challenge and this work, like that of the National Ecosystem Assessment (NEA)[2], explores new scientific territory. The report demonstrates potential approaches, using Countryside Survey and other national scale data, for assessing the state and trends in ecosystem services and the factors impacting upon them.

- The analytical approach to integrated assessment consisted of four phases: 1) In discussion with an expert steering group, a look-up table was agreed that enabled translation of basic measurements (i.e. biophysical variables) recorded in CS into indicators of ecosystem services as defined in the NEA and the Millennium Ecosystem

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[1] In this report indicator is used simply to describe those biophysical variables that have an agreed link to various ecosystem services. Where indicators are already established and officially recognised, for example those species used in Common Standards Monitoring, this is denoted by a capital 'I' as in 'Indicators'. Indicator is also defined in the Glossary.
[2] http://uknea.unep-wcmc.org/

Assessment (MA); 2) Based on these relationships, CS data were then used to quantify the status in 2007, and recent change in biophysical variables that could in turn be interpreted in terms of specific ecosystem services; 3) Greater understanding of the possible causes of change in ecosystem services was then gained by applying explanatory variables to the change in CS variables; and 4) In two cases, statistical models of ecosystem service-related variables were developed and applied to future scenarios of environmental change.

- The approach therefore sought to quantify current status, understand and quantify past change in selected ecosystem services, and use this understanding to explore possible futures.

Overview of trends in ecosystem services at the GB scale

- Indicators analysed: 38 biophysical variables measured in CS were identified as potential indicators of ecosystem service provision relating to different aspects of landscapes and ecosystems: 11 for headwaters, 3 for ponds, 8 for soils, 10 for wild species diversity and 6 for cultural aspects of landscapes. Indicators covered the range of Millennium Ecosystem Assessment service categories.

- Measuring current status and trends: A traffic light system was employed to summarise change in indicators over the time series. Trends were classed as: stable, improved or declined and refer mainly to the 1990 to 2007 period although a number of soil and plant species compositional indicators date back to 1978.

- Different indicators for any one ecosystem service were not always in agreement, suggesting a 'bundle' of biophysical measurements may be required to fully assess ecosystem service status. Using this approach, an overall trend was identified for indicators classified by ecosystem compartment (soils, vegetation, waters, habitat extent and landscapes) and by ecosystem service category.

- Stable or improving: Indicators linked to freshwaters and soils were stable or improving. As they underpin many regulating and supporting services these service categories were also generally stable or improving.

- Declining: Plant diversity indicators were declining (an 8% reduction in number of species in 200m^2 vegetation plots between 1978 and 2007) as was nectar plant diversity, used here as an indicator of one aspect of the regulating service of pollination. Cultural indicators linked to plant species diversity and landscape had also declined.

- Following is a summary of the more detailed analysis of trends observed for specific services linked to freshwaters, soils, plant diversity, pollinator food plants and an exploration of the use of CS data to assess cultural services and interactions between services.

Freshwater quality in headwater streams

- Indicators analysed: Countryside Survey headwater stream biological water quality (representing the ecosystem service of clean water provision) and Community Conservation Index (representing biodiversity) were based on macroinvertebrate data recorded in 249 1km squares in 1990, 1998 and 2007.

- Measuring current status and recent change: Improvements were detected in indicators of biological water quality and biodiversity across GB. Indicators showed improvement in England with mainly no change in Scotland and Wales. In the southern and eastern lowlands of England for example, the estimated percentage of headwater streams in adequate condition has steadily increased from 15% in 1990 to 25% in 1998 and 29% by the time of the 2007 survey.

- Understanding the causes of recent change: There was clear evidence of **spatial** relationships between several environmental characteristics (represented by CS measures from the terrestrial and headwater streams surveys) and ecosystem service indicators. These demonstrated negative effects of intensive land-use, water quality and channel modification on ecosystem service indicators but positive effects of woody cover alongside headwater streams and stream bed substrate diversity.

- Analyses of **temporal** change between 1990, 1998 and 2007 showed fewer relationships between CS measures and headwater stream biological water quality/biodiversity. Increases in cover of Improved Grassland across the 1km square was associated with decreases in biological water quality which is consistent with the finding from the spatial analysis of negative effects of intensive land-use. Increases in streamside cover of woody vegetation were associated with increases in biological water quality again consistent with findings from the spatial analysis.

- A clear trade-off has been identified for streamside woody cover which is favourable for in-stream macroinvertebrates but unfavourable for streamside plant species diversity, nectar plant diversity and appropriate diversity.

Topsoil carbon

- Indicators analysed: Soil samples were taken from the top 15cm of soil in each randomly located vegetation plot (Main Plots) in the same locations within the same 1 km squares in 1978, 1998 and 2007. Topsoil carbon was measured by loss-on-ignition and treated as an indicator of the regulating service of soil carbon storage and climate regulation.

- Measuring current status and recent change: There was no overall mean change in topsoil carbon density between 1978 and 2007. Three significant changes were detected within habitat types. Carbon density increased in Neutral Grassland by 4.7%, and in Bracken dominated vegetation by 14% between 1978 and 1998 and decreased in Arable and Horticulture by 10.7% between 1978 and 2007.

- Understanding causes of recent change: Spatial patterns of change in topsoil carbon density between 1978 and 2007 could be partly explained by changing soil pH, and climate (temperature and rainfall). A drop in sulphur deposition since the 1970's was associated with an increase in soil pH which in turn was associated with reduced topsoil carbon concentration in some locations. These correlative relationships were consistent with expectation from the literature and provide unique large-scale evidence for the link between air pollution and climate change drivers mediated through soils.

Appropriate diversity

- Indicators analysed: *'Appropriate diversity'* is a new term coined during this project. It is mainly a measure of the abundance of 'Common Standards Monitoring' (CSM) indicator plant species in Countryside Survey vegetation plots in 1998 and 2007. These species help indicate how the nature conservation value of habitats varies from place to place and thus help quantify delivery of a cultural ecosystem service. These species can be either 'positive' (indicating a species which is appropriate and valued in the habitat) or 'negative' (a species which is undesirable for that habitat).

- Measuring current status and recent change: Most Broad Habitats showed a general reduction in plant species diversity in CS plots between 1998 and 2007. This was associated with statistically significant decreases in mean species richness of both positive and negative indicator species consistent with the longer term reduction in overall plant species diversity in CS plots between 1978 and 2007 (1.4 fewer plant species were found in 200m^2 plots in 2007 than in 1978).

- Examples of changes include an improvement in appropriate diversity indicated by a significant decrease in negative CSM Indicator cover in three Broad Habitat types; Neutral Grassland, Bog and Dwarf Shrub Heath. There was also a decline in appropriate diversity in the two linear Broad Habitat types (Rivers and Streams and Boundary and Linear Features).

- Understanding causes of recent change: Climate warming since 1980 was associated with a decline in appropriate diversity in Neutral Grassland, Boundary and Linear Features and Rivers and Streams. Deposition of reduced nitrogen also resulted in increased negative indicator cover in a number of Broad Habitats. However, other observed correlations between potential drivers and change in negative and positive CSM indicator richness were often inconsistent with expectation.

- The lack of finely-resolved data on the history of management impacts linked to agri-environment scheme prescriptions persists as a major obstacle in quantifying the effects of habitat maintenance and restoration efforts. Because of this inability to measure positive management, attribution analyses may over-represent negative impacts purely because of insufficient explanatory data availability.

- Modelling possible futures: An example was produced of an established but rapidly developing Europe-wide approach to modelling the impact of multiple drivers on biodiversity. Two niche models were developed for peat-forming mosses called *Sphagnum* in Britain, which is a positive indicator of appropriate diversity for bogs, an extensive habitat prioritised for it's conservation value.

- Scenarios of predicted climate change and atmospheric pollutant deposition impacts (sulphur and nitrogen) were applied to upland peats across the UK to simulate potential change in habitat suitability between 2020 and 2050 for *Sphagnum*. Despite high uncertainties on model parameters, scenario induced changes were extremely small relative to other sources of variation in the predictions.

- A much larger range of presence-based niche models now exist enabling most CSM species to be modelled in a similar fashion. This growing modelling capability allows identification of areas at risk to be identified and thus targeted for intensive monitoring or management measures.

Pollination; nectar plant diversity

- Indicators analysed: The number of nectar-providing plant species in each CS vegetation plot was used as an indicator of potential pollination service delivery in British habitats. Analyses focused on nectar plants for bees (bumblebees and solitary bees combined).

- Measuring current status and recent change: The highest mean numbers of nectar plants per plot in Broad Habitats in 2007 were found in; Calcareous Grassland (12 in 200m^2 Main Plots), Rivers and Streams (5 in 10 m^2 Linear Plots, Boundary and Linear Features (6 in 10 m^2 Linear Plots), Neutral Grassland (4 in 200 m^2 Main Plots) and Broadleaved Woodland (3 in 200 m^2 Main Plots), (note that CS does not provide representative coverage of coastal nor urban habitats).

- Most changes (declines between 1990 and 2007) occurred in small semi-natural habitat patches embedded in common Broad Habitats and on stream and ditch banks. Declines were largest in Arable and Horticulture, Neutral Grassland, Broadleaved, Mixed and Yew woodland and Coniferous Woodland, which all lost on average 1 species in the sample of $4m^2$ plots in the 17 year period. Few changes were detected in larger areas of common Broad Habitat between 1990 and 2007.

- Understanding causes of recent change: Drivers of recent change were found to be habitat specific. Succession[3] was important in suppressing nectar plant diversity between 1990 and 2007 on Rivers and Streams and in woodland Broad Habitats. Boundary and Linear Features showed the opposite effect where successional change in the 17 year period favoured more nectar plant species per plot.

- When vegetation was characterised by species typical of less productive conditions, habitats on infertile soils tended to lose nectar plant diversity while habitats on fertile soils gained nectar plant diversity per sample plot.

- A strong negative correlation between nectar plant diversity and sheep density was detected in upland Dwarf Shrub Heath and to a lesser extent in Bog and Fen, Marsh and Swamp. Therefore higher sheep numbers were associated with lower nectar plant numbers.

- No signals of climate change or pollutant deposition were detected for any habitat.

- Modelling possible futures: Statistical models of spatial variation in nectar plant diversity for bees and butterflies across British Broad Habitats were constructed. The best fitting models included Broad Habitat, % woody cover, climate variables, nitrogen deposition, length of linear features and other landscape attributes as explanatory variables. These models were used to produce predictive maps of nectar plant diversity across Britain and to test the impact on nectar plant diversity of the Defra "environment-only" scenario of agri-environment scheme impacts in English Severely Disadvantaged Areas.

- The modelling work was highly novel and showed much potential for exploring multiple impacts of human activities across British ecosystems. Similar models have been constructed for CSM Indicator species and an indicator of above-ground Net Primary Production.

Charismatic landscapes

- Indicators analysed: '*Charismatic landscapes'* is a new term coined in this project to describe landscapes that people find aesthetically pleasing. Previous research showed that water, wooded features and

[3] The process whereby lack of biomass removal allows the establishment of successive suites of taller plant species eventually resulting in scrub and woodland.

variable relief were associated with such landscapes in selected study areas of Britain.

- These attributes were quantified using CS data on habitats and landscape features to test preliminary spatial expressions of 'cultural services' across England. Relationships between National Character Areas (NCAs) and CS Land Classes were examined.

- <u>Future potential:</u> Results showed that CS data can be integrated with qualitative survey data to provide measures of cultural services; in this case aesthetic appreciation of landscapes. Maps of these services can be made by extrapolating CS 1km square data across GB using the existing Land Class stratification that underpins the CS sampling design. Preliminary analyses showed that CS data has potential for use by policy makers wishing to understand relationships between landscape variables and cultural appreciation of the landscape.

Interactions between ecosystem services

- Analyses of spatial and temporal correlations between ecosystem service indicators were carried out to determine which ecosystem services decrease when others increase, and to what extent these patterns are explained by large-scale ecological gradients across Britain.

- Food production (correlated positively with high above-ground net primary production) and topsoil carbon storage defined the opposing ends of a primary axis of potential ecosystem service provision across 1km squares in Britain. Fundamental ecological constraints on ecosystem productivity put a limit on the extent to which both food production and topsoil carbon storage can be maximised within the same 1km square.

- Plant biodiversity (mean number of species per habitat patch in each 1km square) was highest in the middle of the productivity gradient. This is where nectar plant diversity was also highest. Thus plant diversity was not positively associated with food production or soil carbon storage within plots.

- High freshwater biological water quality was associated with high topsoil carbon, low productivity and high biodiversity. Protection for conservation purposes (SSSI designation) showed an association with high topsoil carbon habitats and soil invertebrate diversity in some habitats. It was less obvious that there was any benefit for terrestrial plant diversity *per se* although the analysis was preliminary in nature. It does suggest that management for conservation might have beneficial effects on provision of other less obvious services.

- Variation in most ecosystem service indicators between plots was highest in the most species-rich 1km squares suggesting that landscapes with diverse habitat mosaics offer the greatest potential for combined delivery of ecosystem services by mixed land-use across the 1km square.

- Future potential: Values of ecosystem service indicators can be predicted in terms of the proportion of Broad Habitats present in a 1km square. This offers the prospect of model-based mapping of a range of ecosystem service indicators across all 1km squares in Britain using the new Land Cover Map for 2007 in combination with the ordination models initially developed in this project.

Understanding observed changes in ecosystem services

- The most substantial recent transformations of the British landscape and therefore of potential ecosystem service provision probably happened prior to 1990 during the post-WWII period of urban expansion and agricultural intensification. By 1990 spatial relationships between anthropogenic drivers and many ecosystem service indicators were often already set in place. This applies for example to sheep density and nectar plant diversity as well as nitrogen deposition and a range of plant diversity indicators[4].

- There are however, indications of ongoing changes linked to driving forces still active across British ecosystems. For example, recent changes in indicators of appropriate diversity are partly explained by climate warming and nitrogen deposition while changes in headwater stream quality are linked to increased woody cover and the extent of surrounding intensively managed grassland. In addition, changes in topsoil carbon density are correlated with recent climate warming and reductions in sulphur deposition working via correlated increases in soil pH. These new results are inevitably based on generally weak signals given the large amount of variability typical of CS surveillance data. Yet they constitute novel and unique large-scale evidence that key global change phenomena have left their footprint on the delivery of ecosystem services across British ecosystems and have continued to drive change in the last two decades.

- While statistically significant changes in ecosystem service indicators are often detected between 1990 and 2007, further research is required to evaluate the impacts of these changes. For example, the reductions in nectar-plant diversity between 1990 and 2007 in four Broad Habitats were substantial enough to move the mean richness for each habitat closer to the mean value for a different more species-poor habitat type. Yet, the functional significance of these changes for pollinating insects and pollination, requires calibration against studies that have tracked in

[4] See Fowler *et al.* (in prep). Report on Trans-boundary Air Pollution. See www.rotap.ceh.ac.uk.

detail the dynamic linkages between declining nectar sources and pollinating insects.

- Despite uncertainty over the importance of recent changes in ecosystem service indicators, future delivery of biodiversity-linked services is set against a context of terrestrial plant species diversity reductions in the wider countryside since 1978. Most of this decline remains unexplained. One plausible explanation is lagged responses to earlier 20th century habitat fragmentation. A second is that disturbance regimes have changed. Post-WWII rural depopulation and agricultural intensification have led to reduced disturbance in many habitats in the countryside, including marginal land in the lowlands, linear features and lowland broadleaved woodland, and increased disturbance on more productive land. The net effect would be that optimum conditions for plant diversity are less prevalent in the wider countryside.

- Whether further advances can be made in attributing observed changes in CS depends on further efforts to acquire and apply finely-resolved explanatory variables. Land management has been the dominant influence on habitat extent and condition but in recent decades intensive management has undergone a targeted reduction as a result of agri-environment schemes. A key issue in detecting the positive outcomes that would be expected is the lack of high resolution agri-environment management data. Despite putting considerable effort into acquiring GB-scale spatial datasets on agri-environment scheme status, such data still do not exist in a coordinated spatially coherent format that allows coupling of polygon-level ecological responses with detailed information on current and past management impacts under agri-environment schemes. The risk is that if ecological responses cannot be easily coupled with datasets that track positive management impacts, large-scale attribution results from CS will be biased toward expressing the effects of negative drivers. All similar surveys will have to contend with this same limitation on the ability to demonstrate positive policy outcomes.

Good and bad news for ecosystem service delivery

- Ecosystem service indicators based on CS data suggest bad news for terrestrial plant species diversity which showed more declines than increases in CS data. This goes against current biodiversity targets. In contrast there is good news for headwater stream biodiversity and water quality indicators which showed a stable or improving situation across Britain. In addition, topsoil carbon density has shown very little change since 1978 suggesting no major loss of carbon to the atmosphere. Further work is needed to provide assessments of equivalent detail for other ecosystem services and their indicators; for example, to include above-ground carbon stocks and other indicators relating to supporting services such as primary production. The techniques and approaches used in this report can be readily applied to other indicators in future integrated assessments.

- Despite the bad news for plant diversity, new agri-environment schemes may have sufficient geographic penetration to make a major difference to biodiversity-related ecosystem service provision across British ecosystems. The widespread uptake of agri-environment schemes is aimed at promoting delivery of mixed ecosystem services at the farm scale but some of these rely on the existence of residual species pools that are sufficiently diverse to enable community reassembly given reduced management intensity. Evidence from the integrated assessment and elsewhere suggests that eutrophication and succession have been potent forces in depleting local species pools in the last 30 years. However, there is also evidence that in places sufficient resilience remains for local extensification measures to produce a positive increase in desirable species abundance.

- Succession emerges as an especially important driver of reductions in nectar plant diversity and appropriate diversity but also appears to have had a positive effect on headwater stream biological water quality. The extent to which joint delivery of these apparently competing services is promoted requires local management solutions based on evaluations of the importance of conserving existing species pools for pollination and cultural enjoyment versus protecting headwater biodiversity and water quality from nutrient surpluses. Analysis of the interrelationships between ecosystem service indicators across Britain suggests that the endpoints of local strategies designed to maximise ‘bundles’ of ecosystem services will be ultimately constrained by broader landscape-scale trade-offs between productivity and species richness that reflect over-arching soil and climate gradients. The greatest room for ecological manoeuvre in maximising mixed services via management and thus also possibly resilience for uncertain futures seems to be in those 1 km squares typified by a high diversity of habitats and species and where, on average, productivity is intermediate.

- This work leads the way in developing integrated assessment approaches for assessing changes in ecosystem services. The results are feeding into NEA and together they can be used to inform future monitoring requirements and scenario testing to help plan for more sustainable use of our natural capital.

Chapter 1: Introduction and overview of ecosystem service indicator status and change

L.R. Norton, L.C. Maskell, B.A. Emmett, P. Scholefield

- Countryside Survey provides a unique time series of systematically collected data which incorporates measures of soil, water, vegetation and landscape taken at the same locations. This dataset provides excellent opportunities for the development of exploratory science integrating data collected using a sampling framework which enables reporting at the GB[5] scale.

- Increasing pressure on natural resources and concerns about environmental change has led to a re-casting of natural and semi-natural ecosystems in terms of human requirements both currently and in the future. Termed '***ecosystem services***', these provide a focus for policy makers seeking to ensure sustainable development.

- Ecosystem services provide a significant challenge for scientists, not least in framing and defining what they may constitute. Beyond definitions is the challenge of how to measure services and understand how they may interact in any given ecosystem. Countryside Survey data provide a unique opportunity to investigate how to measure ecosystem services and interpret changes over time.

  The work described in this report uses data collected for Countryside Survey alongside other relevant data sources to provide '***integrated assessments***' of a range of measures of ecosystem services at the national scale. Understanding ecosystem complexity at any scale is a significant scientific challenge and this work, like that of the National Ecosystem Assessment (NEA)[6], explores new scientific territory. The report demonstrates potential approaches, using a range of Countryside Survey and other national scale data, for assessing the state and trends in ecosystem services and the factors impacting upon them.

[5] GB (Great Britain) refers to England, Wales and Scotland only while the UK includes Northern Ireland. Although the Northern Ireland Countryside Survey contributed data to the reporting of UK-wide results in 2008, the time series of soils, waters, vegetation and habitat cover data only apply to GB. Hence GB and not the UK form the limit of the sampling domain from which data were analysed for this report.
[6] http://uknea.unep-wcmc.org/

Ecosystem Services	***Box 1.1***
A collective term to describe ecosystem outputs, functions and processes which have human beneficiaries. These include a range of intermediate services which may be involved in **regulation** (e.g. flood control), **support** (e.g. nutrient cycling) or **provisioning** (e.g. food and fresh water) of ecosystems. These services are essential for maintaining conditions for life on earth (Fischer and Turner 2008).	

1.1 Background

An evolving scientific and policy context

Measuring ecosystem services, understanding the interdependencies between underlying drivers and ecosystem services and valuating ecosystem services have become major scientific challenges (Carpenter *et al*. 2009; Feld *et al.* 2009; Norgaard 2009). The launch of the Millennium Ecosystem Assessment (MEA)[7] in 2003 illustrates the considerable intellectual development surrounding the understanding of ecosystem goods and services. Since then, interest has grown in refining the concepts and their implementation at various scales; ecosystem services are central to the EU Framework 7 programme and emerge in many national initiatives. In the UK, Defra funds an ongoing programme of research on these issues with the aim of refining definitions and assessing the feasibility of using such a framework for the assessment of stocks and trends of natural resources[8]. The UK National Ecosystem Assessment is the first analysis of the natural environment of the UK in terms of the benefits it provides to society and its continuing economic prosperity. The National Ecosystem Assessment (NEA) is being carried out as part of the Living With Environmental Change (LWEC) programme, which commenced in mid-2009 and will be reporting in early 2011.

The policy framework surrounding land use and management in the UK has evolved greatly since the reporting of the Countryside Survey in 1998. In particular, we have witnessed the increasing importance of new strategies promoting sustainable development at the EU and UK levels including: 'The Economics of Ecosystems and Biodiversity (TEEB) for policy makers[9]; 'Securing a healthy natural environment: An action plan for embedding an ecosystems approach[10]; 'Sustainable Development Action Plan 2009-2011[11] and the recent 'Food 2030[12] (the first UK food strategy released for 50 years). These changes have been influenced by and have introduced new

[7] http://www.millenniumassessment.org/en/Global.aspx
[8] E.g. NR0107 'England's Terrestrial Ecosystem Services and the Rationale for an Ecosystem-based approach.
[9] The Economics of Ecosystems and Biodiversity http://www.teebweb.org/ForPolicymakers/tabid/1019/language/fr-FR/Default.aspx
[10] http://www.defra.gov.uk/environment/policy/natural-environ/documents/eco-actionplan.pdf
[11] http://www.defra.gov.uk/sustainable/defra/pdf/sd-action-plan-2009-2011.pdf
[12] http://www.defra.gov.uk/foodfarm/food/strategy/index.htm

terminologies and classifications of our environment which attempt to capture the multiple benefits derived from land and water. Among the terminologies and classifications, the concept of 'ecosystem goods and services' (Table 1.1, also ***see glossary*** for definitions of these and related terms), used to describe all the benefits which humans derive from the natural environment, is particularly relevant for the measurements made by Countryside Survey. The need for good quality evidence to contribute to policy making and sound decisions in the area of ecosystem services is outlined in the recent 'Defra's Evidence Investment Strategy'[13]. Countryside Survey uses a sampling framework designed for reporting at a GB scale and provides a unique time series of data which incorporates measures of soil, water, vegetation and landscape taken taken simultaneously at fixed locations for each survey. The Countryside Survey dataset is therefore uniquely capable of helping to quantify current status and recent trends in ecosystem service indicators and to better understand the extent to which spatial and temporal changes in one service may happen in parallel with increased or decreased provision of others.

CEH committed to develop integrated analysis of ecosystem services from rural landscapes in its Science and Innovation Strategy Board Q1 theme on sustainable landscapes *(C2.1. Integrated assessment of trends in ecosystem resources and services (2006-2010)).* This proposed that the next Countryside Survey (2007) would deliver an ecosystem assessment of GB, going well beyond the Millennium Ecosystem Assessment standards and ensuring compatibility with developments being led at the European level. This programme has now been updated within the new CEH science strategy 'Integrated Science for Our Changing World'[14] with an emphasis on integrating science across disciplines. Challenge II of the new CEH strategy promises to identify processes linking physical and chemical changes in the environment with ecosystem responses whilst Challenge IV undertakes to quantify the impact of environmental change on natural resources. In a similar way, the Living With Environmental Change (LWEC) programme focuses on the development of the interdisciplinary evidence base, tools and processes that are needed to inform public and policy debates, individual choice along with whole-system assessments and risk-based predictions of environmental change and the effects on ecosystem services.

Countryside Survey is a key dataset for integrating data across natural science disciplines and developing ways of answering high level questions concerning the interdependencies between ecosystem services and underlying drivers of change. Additionally, for some data variables, Countryside Survey provides opportunities for understanding the potential impacts of drivers of change at the GB scale.

[13] http://www.defra.gov.uk/evidence/science/how/documents/eis-100126.pdf
[14] http://www.ceh.ac.uk/science/corpstrat.html

Scientific challenges

Although the concept of ecosystem services is intended as a step towards sustainable management of natural resources, many challenges remain for scientists working in this area (Carpenter et al 2009). Key questions include:

- How do we link changes in ecosystem services and ecosystem functioning to changes in human well being?
- How should biophysical measurements, ecosystem functions (the mechanisms by which services are generated) and services be defined?
- What are the critical ecological elements underpinning the sustainable provision of a service?[15]
- How does biodiversity influence ecosystem functions and services (see Anton *et al.* 2010)?
- At what scales are services generated and how do we evaluate and map services at those different scales?
- What are the interrelationships, synergies and trade-offs between services[2]?
- What are the data limitations and how can they be addressed?

The concept of ecosystem services is a valuable one as it makes the link between natural resources and human well being. The science required to provide underpinning evidence for these complex links is, however, in its infancy. The predominant ‘culture’ in science, influenced in part by funding mechanisms, is to focus on narrow questions for which it may be possible to find an answer using short-term research projects. Inevitably these projects fail to incorporate the inherent complexity of both natural and managed ecosystems and often provide only limited context-specific information (see Carpenter *et al*. 2009, Nicholson *et al*. 2009).

Countryside Survey goes some way towards providing the kind of long-term, large-scale integrated dataset which will be essential for a scientific understanding of effective management for ecosystem services. The limitations of the Countryside Survey dataset are largely due to: limits on time and resources when surveys took place, the rationale which shaped the specific surveys at the time of their initiation and the temporal structure of the surveys. Inevitably there are ‘gaps’ in the data in relation to ecosystem services, e.g. those resulting from the limited soil and water sampling in squares, or from timing of surveys. Potentially, the most significant gaps are the lack of detailed land management data and relevant social data at the survey square level, (although land management data are becoming increasingly available). Countryside Survey is one of few such datasets available for integrated environmental analysis and its wider use alongside external datasets may be part of the shift in the ‘culture’ of science, which is necessary to ensure the greater degree of integration both across the natural sciences and spanning natural and social sciences, and which will be required for an understanding of ecosystem services.

[15] Taken from summary of presentation by Dr Phil Warren at BES-Defra workshop on ecosystem services

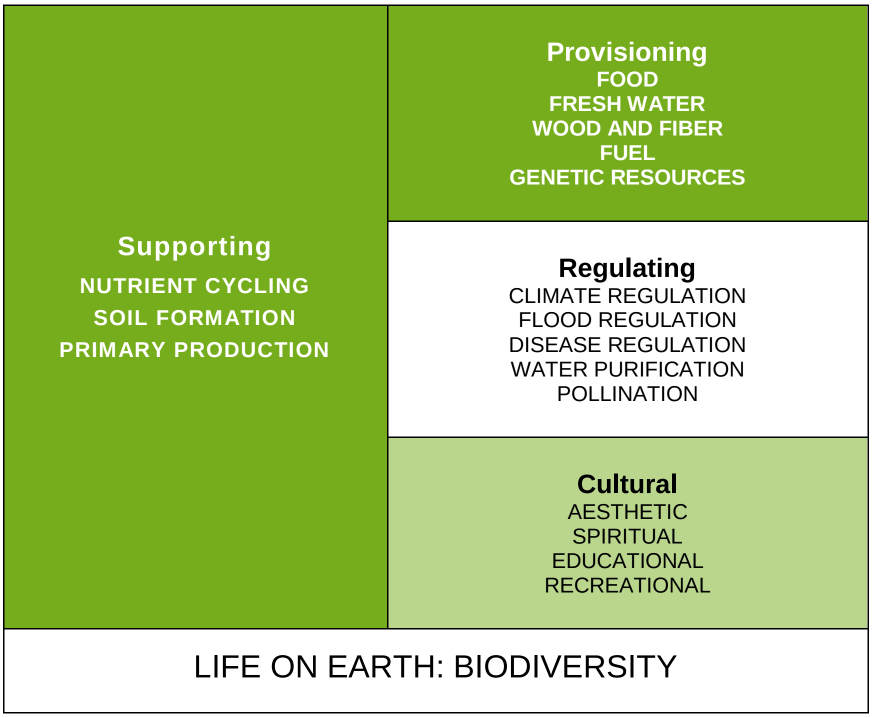


Figure 1.1: Ecosystem services are the benefits people obtain from ecosystems. These include provisioning services such as food and water; regulating services such as regulation of floods, drought, land degradation, and disease; supporting services such as soil formation and nutrient cycling; and cultural services such as recreational spiritual, religious and educational benefits.

The report demonstrates ways to quantify or measure ecosystem services, but it does not make any attempt to ***value*** the ecosystem services which we report on. This requires different expertise to that required for measuring services. Service valuation is a particular area of contention in work on ecosystem services. This is reflected in a lack of agreement among ecologists and economics on definitions relating to ecosystem services (Boyd and Banzhaf 2007). For example, the word ‘service’ may be used to apply to something that can be directly measured, e.g. crops, or to something such as nutrient cycling which is a supporting process, yet nutrient cycling is not a material end-product and is difficult to measure. Wallace (2007) suggests the service arises at the point at which an ecosystem directly provides an asset that is used by one or more humans. However, the components that contribute to a service, e.g. water quality which affects fish populations which provide a service required for angling, may be a service in their own right, e.g. for drinking water, water quality is the service. This area is of key importance to policy makers seeking to ensure that the true value of ecosystems and the services they provide are taken into account in policy decision-making[16].

[16] ‘An introductory guide to valuing ecosystem services’ http://www.defra.gov.uk/environment/policy/natural-environ/documents/eco-valuing.pdf

Another area of complexity is that services require consumers whose locations and levels of demand may vary geographically, socially, and economically. Additionally, the consumption of one service may have implications for other services which in their turn affect different groups of consumers, e.g. removal of timber in a catchment may be a provisioning service for one group of people but result in changes to another service, water flow regulation, to those further downstream. For services such as the provisioning of genetic diversity the consumer is assumed to be represented by future generations. Valuing ecosystems requires different expertise to that used for the integrated assessment and will require collaboration with scientists outside of CEH to look at the social and economic aspects of ecosystem services. The glossary in this report seeks to clarify terminology used in this report and as far as possible to ensure consistency with terminology used elsewhere.

Finally, the limitation of our abilities to define and measure ecosystem services also leads to problems in uses of surrogates, metrics or descriptors of ecosystem services. During this study, two new terms were coined as descriptors of ecosystem services. *'Charismatic landscapes'* is used to encapsulate several variables describing cultural ecosystem services. *'Appropriate diversity'* is used to describe a particular measure of plant diversity which represents a cultural ecosystem service.

1.2 Approach

The potential uses of the Countryside Survey dataset in determining and understanding ecosystem services are many and varied. Table 1.2 summarises how Countryside Survey biophysical measures have been related to, and used to quantify, ecosystem services. Recent trends in service provision are also summarised for GB and the individual countries for the ecosystem services covered in this report. A complete table of ecosystem services and their indicators is provided in Appendix 1.1 (it should be noted that both tables involve an element of expert judgement). In each of the chapters focusing on specific ecosystem services, an ecosystem service cascade (*as in* Haines-Young and Potschin 2007) has been used to indicate how biophysical measures from Countryside Survey may be linked to ecosystem functions and services (Fig. 1.2). This approach places the work in this report in the context of other work on ecosystem services and of previous reporting on the ouputs of Countryside Survey. For example, results from the Headwater Streams Report (Dunbar et al 2010), the Soils Report (Emmett et al 2010) provide the evidence base for the much of the status and change sections in chapters 2 and 3 and the response variables whose patterns of change are analysed in the attribution sections of those chapters. The various country and UK level main reports as well as reports from past surveys and the many scientific papers that have utilised CS data, have also provided a rich foundation of evidence that the Integrated Assessment has drawn on to quantify and communicate the potential causes and significance of change in Ecosystem Services across Britain.

However, this report should be seen as the start of an ecosystem service-led research agenda and not a final and comprehensive assessment of all services and their drivers of change. This is because it has only been possible to carry in-depth analysis of a limited number of a wider set of ecosystem services given the time and resources available. The emphasis has been on developing the best possible scientific approaches so that concepts and methods described here can be applied to a wider range of ecosystem services at a later date. Figure 1.3 summarises the report content and indicates the extent to which each of the services has been explored. It is hoped that the science in this area, together with its potential policy applications will be explored more fully in time. This report demonstrates the potential for Countryside Survey to be used for work on ecosystem services by:

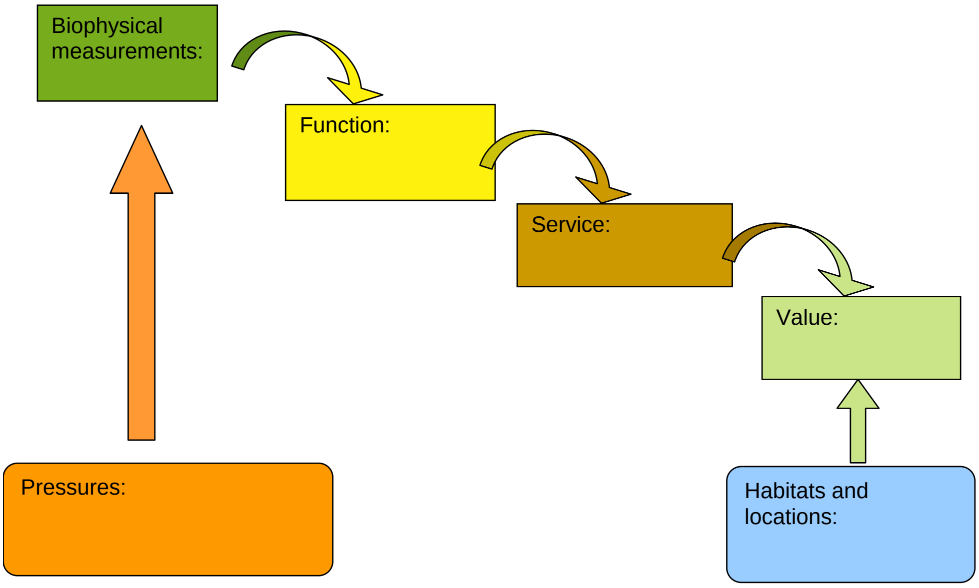
1) linking biophysical measures from Countryside Survey to ecosystem services using pre-existing methodologies[3] to provide current ‘service’ status,

2) piloting approaches for retrospectively evaluating impacts of drivers on example ecosystem services (attributing change – Box 1.2) and 1.3) exploring trade-offs between ecosystem services,

3) using statistical models to generate simulations of future changes in ecosystem service delivery given scenario-driven ecological change.

Maps have been used throughout the report to provide visualisations of the status, changes and potential changes in ecosystem services. A brief explanation of mapping techniques used in this report, which should be used in conjunction with the maps included in the service chapters is available in Box 1.3.

Figure 1.2: The ecosystem service cascade (after Haines-Young and Potschin 2007).



Attributing change - Interpretation of results Box 1.2

It is very important to note that relationships have been defined between potential drivers and response variables through iterative empirical model building. Despite the fact that it is convenient language to say “x influences y”, as this is an observational study, there may be multiple reasons why this might be:

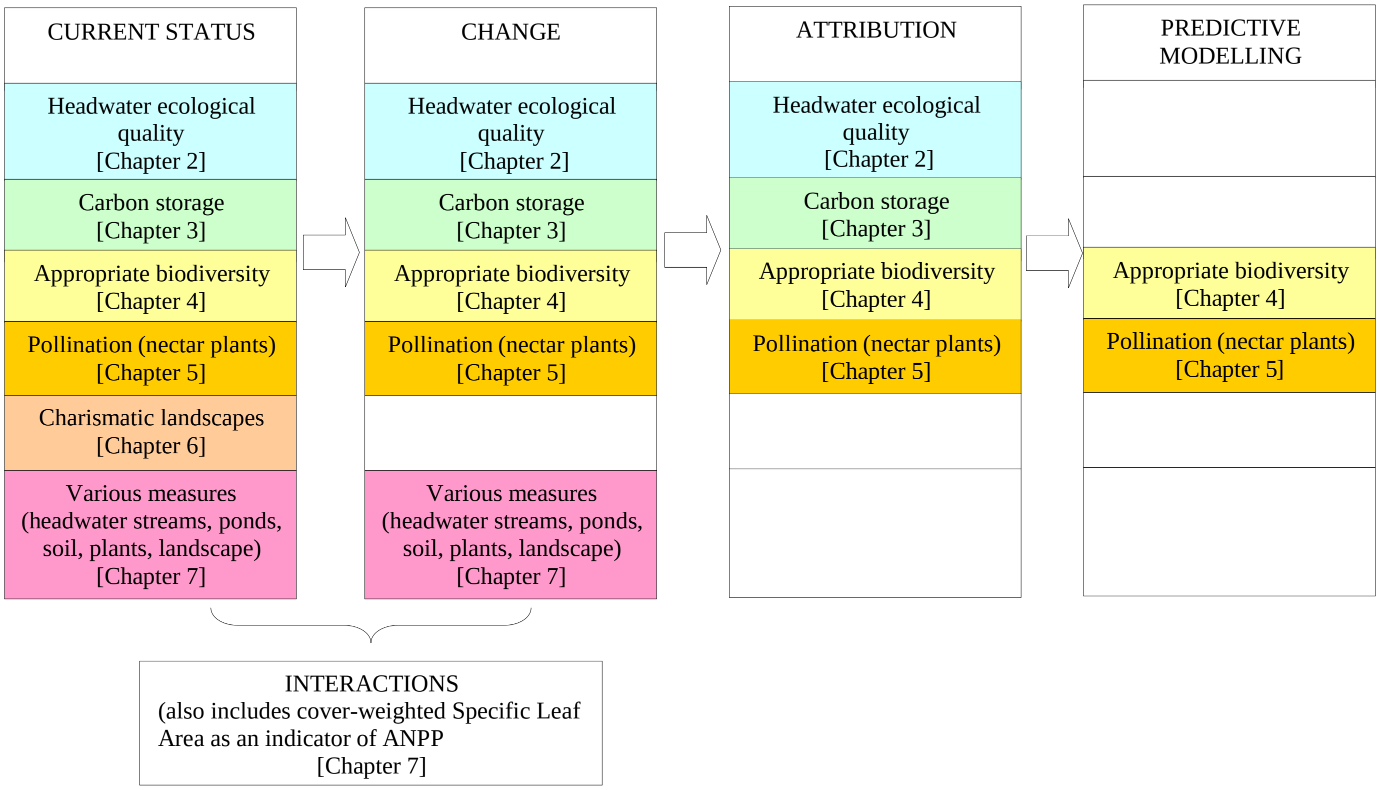
- x could cause y;
- y could cause x (in this study, it is fairly certain in most instances that this is not a possibility);
- x and y may interact with each other (e.g. percentages of an inclusive set of different categories (such as a set of land cover types) are constrained to 100%);
- any relationship between x and y may be pure chance, this is controlled for by using multiple lines of evidence for selecting variables to include, and avoiding testing large numbers of potential variables and models;
- x and y may both be effects of another variable (in this study this is often possible).

Table 1.2: Summary of natural capital and ecosystem service indicators reported by Countryside Survey in 2007. Colours indicate trends in service: Green (in white text) = improved, amber (in light grey text) = stable, red (in bold text) = declined). No colour highlights indicators that exist or can be derived from CS but their application needs further development.

MA category	Service	Ecosystem component measured in Countryside Survey			
		Waters	Soils	Plant diversity	Landscape
Biodiversity	Wild species diversity	Water plant species richness; Headwater Habitat Quality Assessment Score; Number of ponds	Soil phosphorus availability index; soil pH; soil N content		Increase in Broad Habitat of: Broadleaved Woodland; Improved grassland; Neutral Grassland; Acid Grassland
		Number of scoring macroinvertebrate families; Macroinvertebrate species richness; Habitat Modification Score; Community Conservation index for Macroinvertebrates			No change in Broad Habitat area of: Coniferouse Woodland; Dwarf Shrub Heath; Fen, Marsh and Swamp; Bog
		Pond Plant species richness	**Soil invertebrate diversity**	**Appropriate plant diversity; Butterfly nectar sources; Bee nectar sources; Lowland farmland bird plants; Number and cover of invasive and non-**	
		Phosphorus load		Richness of crop wild relatives	
Provisioning	Food and Fiber	Phosphorus load	Total soil invertebrate count; Soil nitrogen content; Soil phosphorus availability index; soil pH		
			Trace metal concentrations in soil		
				Butterfly nectar sources; Bee nectar sources	
			Soil nitrogen availability index	Richness of crop wild relatives; Number of notifiable weeds; Plant carbon fixation (net primary production)	
Supporting	Nutrient cycling		Total soil invertebrate count		
			Soil carbon density		
	Soil development		Total soil invertebrate count		
	Acidity buffering		Soil pH		
	Primary production			Plant carbon fixation (net primary production)	
Regulating	Pollination			**Butterfly nectar sources; Bee nectar sources**	
	Water flow regulation	Habitat Quality Assessment Score			
		Habitat Modification Score			
	Water purification	Mean Trophic Rank of water plants	Total soil invertebrate count; Soil nitrogen content; Soil phosphorus availability index; soil pH		
		Average score per Taxon of macroinvertebrates; Overall macroinvertebrate status class; Number of scoring macroinvertebrate families; Acid Waters Indicator Community for macroinvertebrates; Acid Waters Indicator Community for macroinvertebrates	Trace metal concentrations in soil; Soil carbon density		
		Pond Status class			
		Phosphorus load			
	Climate regulation		Soil carbon density	Vegetation biomass as a carbon store; Plant carbon fixation (net primary production)	
Cultural	Cultural			**Lowland farmland bird plants; Appropriate diversity**	**Length of stone walls; Length of hedgerows**
					Charismatic landscapes; Appropriate landscape diversity; Hedgerow condition

Trends up to 2007 are summarised across time intervals that vary in length reflecting differences in the baseline year in which ecosystem service indicators were first recorded as follows. Waters: macroinvertebrate scores (1990), habitat scores and phosphorus load (1998), Pond species richness (1996), number of ponds (1998). Soils: Soil carbon and pH (1978), soil invertebrates, trace metals, nitrogen and phosphorus (1998). Plant diversity: Lowland farmland birdfod plants (1978), all others (1998). Landscape: all variables (1998).

Figure 1.3: The extent to which each ecosystem service has been explored in this report.



1.3 The integrated assessment database

The integrated assessment introduces novel methods of linking the different data collected at the same sites within Countryside Survey. It also links Countryside Survey data to other national scale environmental data at a common 1km scale to enable integrated research. The resulting database includes datasets within broad groupings which include; climate, deposition, hydrology and land use (the full list appears in Appendix 1.1). Not all data are consistent across the whole of GB at a relevant scale. For example, the process has highlighted differences between countries in terms of data holdings on agri-environment schemes (not used in this report). The integrated assessment dataset is as much a product of the integrated assessment as is this report and will continue to be built upon and used for future analyses exploring the Countryside Survey data.

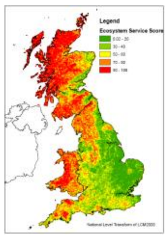
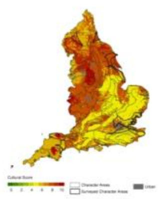
Methods used to map Ecosystem Services based on Countryside Survey data — *Box 1.3*

Introduction
Options for mapping potential ecosystem service delivery in GB are constrained by data availability. Given the data available, various methods can be used to interpolate or scale-up to Great Britain and examples are shown in the chapters in this report and in the appendices. Although maps offer an appealing 'census' description of an area, extreme care needs to be taken in their use. The purpose of this box is to provide information to ensure that the maps in this report are used appropriately. This chapter describes the three key options for mapping (two mapping, one modelling, Table 1.3) used in this report and should be used in conjunction with interpreting the maps presented in the service-related chapters.

Model based approach to mapping ecosystem services
A model based approach to mapping uses observed values of the variable of interest (known as the response variable) together with corresponding information on other variables (known as covariates) to build up a statistical model that can be used to make predictions. Standard statistical models assume that observations of the response variable are independent. When survey data is collected with some inherent spatial element, independence in the observations is rare. This is because observations close to one another are more likely to be similar than observations far away. This is known as spatial autocorrelation. We account for this spatial autocorrelation by including a spatial trend surface that captures any residual spatial variation in the data. This ensures that parameter estimates and their associated standard errors are unaffected by spatial dependence.

Mapping vs. Modelling
The primary aim of mapping is to provide a visualization of the data, while modelling provides predictive capability. The final map product is a combination of both the visualization and underlying model prediction and careful thought is required as to how the map should be interpreted. Generally speaking, the utility of prediction and utility of visualisation in maps tend to be inversely related. The example maps below are provided primarily to give examples of how the distribution of ecosystems services can be visualized and it should not be assumed that the values represent an appropriate prediction of service provision for a specific location.

Three approaches to mapping ecosystem services: model based and direct mapping of land class means.

Approach	Pros	Cons	Assumptions	Example	Example map
Model based	Observations of the response variable are independent. Spatially implicit Uncertainty of predictions and upper and lower confidence bounds are easy to estimate and map. Scenario testing capabilities: reaction of response to changes in the covariates resulting from a specific policy scenario.	Higher level of parameterisation The model may not be ecologically sensible. Can be a time consuming process from building the model, and from selection of the best model for making predictions.	Covariates are independent.	Pollination (Nectar plant diversity (right) – Chapter 5	
Broad Habitat Mapping using census land cover map	Simple and fast production Low level of parameterisation	Less predictive capacity. Insensitive to spatial differences in values that are not explained by Broad Habitat differences.	Values are representative of a wider spatial area	Broad Habitat-based synthesis of Haines-Young & Potschin (2007) matrix	
Land Class Mapping	Uses landclass means derived from square level observations Simple and fast production Low level of parameterisation Can form the basis of predictive models Allows upscaling of fine-grained attributes recorded in sample squares in each land class, for example, length of hedgerows, and numbers of veteran trees.	Spatially distant areas may be ascribed similar values irrespective of local effects Recuced predictive capacity	Values are reopresentative of a wider spatial area	Appropriate Diversity (Chapter 4), Charismatic Landscapes shown here (Chapter 6).	

Chapter 1: Online appendices

1.1: Natural capital and ecosystem services reported by Countryside Survey in 2007 (the extended version of Table 1.2).

Chapter 2: Freshwater quality; clean water provision and biodiversity

M.J. Dunbar, S.M. Smart, J.F. Murphy, R.T. Clarke, R. Baker,

F.K. Edwards, L.C. Maskell, P. Scholefield

Summary

- Provision of clean water is an important ecosystem service. Internationally, it is recognised that environmental change, including climate change, is a major risk to this provision;

- Land management actions (including lack of management) have a clear influence on the ecosystem service of clean water provision (as measured by stream biological quality), and on headwater stream biodiversity. These actions can be detected at a range of spatial scales: in-stream, along-stream and wider landscape;

- Macroinvertebrates (small animals living on the river bed) are a very useful indicator of stream biological quality due to their response to multiple stressors and their ubiquity. Here they are used to represent the ecosystem service of clean water provision;

- The analysis undertaken, separates combined impacts on stream biological quality and biodiversity. For example, one can discern additive effects of extent of arable land, inorganic nutrient concentrations and channel and riparian management. It separates spatial and temporal patterns in the Countryside Survey datasets. Patterns described below are spatial except where specifically noted;

- Extent of arable land in the wider landscape has a negative impact on stream biological quality and biodiversity. Increases in cover of intensive grassland are associated with a decline in status, however it is not clear whether this is related more to intensification of Neutral Grassland or reversion from arable;

- Soluble Reactive Phosphorus concentrations from single spot samples in the stream are strongly related to stream biological water quality and biodiversity, there are likely to be multiple indirect mechanisms for this;

- A measure of historical physical degradation of the stream channel by human activities (the extent of resectioning) is also negatively associated with stream biological quality and biodiversity, but in some cases the effects are difficult to separate from those of Soluble Reactive Phosphorus concentration;

- Increased riparian woody cover along the streams is associated with improved stream biological quality and its increase through time is beneficial: but caution is needed since there are negative associations between woody cover and other Countryside Survey measures such as pollinator species and appropriate diversity; effects on instream vegetation have not been assessed in this study;

- The diversity of the composition of stream channel substrate materials (sands, gravels, cobbles etc) at reach scale (hundreds of metres of stream channel) is associated with improved stream biological quality at the scale measured here (tens of metres).

2.1 Introduction

Background

The countryside of Great Britain has been highly influenced by human activities. The Countryside Survey (CS) 1km x 1km squares provide a representative sample of the major land classes in Great Britain with data collected on both terrestrial and freshwater ecosystems (Carey *et al.* 2008). Terrestrial data includes multiple measures which reflect intensity of land management. Freshwater measures include the sampling of a single headwater stream site per 1km survey square at which a biological sample of the stream macroinvertebrates is taken, the vegetation and physical habitat are surveyed and the water quality sampled (Carey *et al.* 2008).

Streams and rivers provide multiple ecosystem services. Hence information on the relationship between land management activities and the response of stream ecosystems is important if policies for their protection are to be effective.

In addition to the variations associated with the changing productivity of land as one moves from lowlands of the south and east to uplands of the north and west, many of the Countryside Survey measures vary geographically. Further, there were notable changes recorded in the character of vegetation in Streamside Plots from 1990 to 2007 (Carey *et al.* 2008). In particular, there was evidence of increases in cover of woody species. This was matched by some significant changes in the headwater stream quality elements between 1990 and 1998, and between 1998 and 2007 (Carey *et al.* 2008; Dunbar *et al.* 2010a). Hence, Countryside Survey provides an extremely useful dataset with which to examine both spatial and temporal patterns in combined terrestrial and freshwater data.

The overall aim of this study is to explore the integrated assessment of terrestrial and freshwater Countryside Survey data. This exploration is used to establish empirical relationships between a set of potential explanatory variables, hereafter termed CS measures, reflecting intensity of land management, and stream biological quality, representing the ecosystem service of clean water provision. These relationships will also help to inform policies for the protection and management of headwater streams. Potential mechanisms for the observed relationships are discussed.

In this study relationships were examined between selected Countryside Survey measures reflecting land management and activities and headwater stream biological water quality and biodiversity. Analysis was undertaken using Countryside Survey data from 1990, 1998 and 2007. Only stream sites are considered in this study: ditch / drain sites are also surveyed in Countryside Survey where a stream site is not present in the square, but these are reported separately and not considered in this analysis. The analysis was undertaken using a statistical modelling approach, integrating data from the terrestrial and headwater streams components of Countryside Survey.

Three broad types of CS measures were tested for relationships with the spatial and temporal patterns of stream biological water quality and biodiversity:

- **In-stream** measures taken from the headwater stream survey at or close to the biological sampling site (1998, 2007 data only) (e.g. stream nutrient concentrations, the condition of the stream channel and banks);
- **Along-stream** measures from the terrestrial survey measured close to the stream, but not necessarily coincident with the biological sampling site (1990,1998, 2007);
- **Wider landscape** measures, expressed as Broad Habitat cover in the Countryside Survey square, this was used as a proxy for land management intensity (1990, 1998, 2007).

These measures together represent aspects of active land management, or lack of active management which are partly influenced by government policy towards the countryside. They vary temporally (from year to year) and spatially (hundreds of sites ranging over hundreds of km).

Exploratory analysis was also undertaken to examine the relationships between the CS measures.

2.2 Biophysical Measurement

Rationale for selection of target organisms

This chapter focuses on the use of measures derived from the macroinvertebrate samples from Countryside Survey headwater stream sites. Macroinvertebrates are animals visible to the naked eye, such as snails, worms, leeches, shrimps, mayflies, dragonflies, water-bugs, beetles, caddis flies and midges. Some are aquatic for their entire life-cycle, others require water for their larval stages but exit the water as adults. Most live on the river bed, but some are associated with the water surface. Macroinvertebrates have been sampled in Countryside Survey since 1990.

Macroinvertebrates were chosen over macrophytes (higher plants) which are also surveyed during Countryside Surveys for several reasons:

1. The underpinning science of how macroinvertebrates respond to a range of anthropogenic pressures is well developed. Both within and beyond the UK, macroinvertebrates represent the most well studied biological group for biomonitoring purposes, particularly in terms of their utility as water quality indicators (Rosenberg and Resh 1993; Wright *et al.* 2000; Jones *et al.* 2010);
2. Macroinvertebrates are present in almost all streams while macrophytes are often naturally absent from the headwater streams sampled in CS;
3. Data are available from three surveys (1990, 1998, 2007) for macroinvertebrates, rather than two for macrophytes (1998, 2007);
4. There is a well-established procedure (RIVPACS/RICT (Wright *et al.* 2000; Davy-Bowker *et al.* 2008)) for correcting macroinvertebrate sample indices for their natural spatial variation in the UK. The corresponding tool for macrophytes (LEAFPACS (Willby *et al.* 2009)) is comparatively new and less well tested.

Macroinvertebrate biotic indices

Macroinvertebrate community data for each Countryside Survey square for each survey were summarised as biotic indices. Two indices were chosen for examining biological water quality. These are Average (BMWP – Biological Monitoring Working Party) Score per Taxon (ASPT) and number of BMWP scoring taxa. These indices are used as standard indicators of water quality by UK monitoring agencies (Environment Agency, Scottish Environment Protection Agency and Northern Ireland Environment Agency). They are appropriate to use with the RIVPACS sampling protocol employed to collect macroinvertebrate data in Countryside Survey. Values for the scores were expressed as an Observed ÷ Expected (O/E) ratio (see Appendix 2.1 for further details), hereafter, all references to Average Score per Taxon and number of BMWP scoring taxa refer to their O/E ratios.

The Community Conservation Index, or CCI (Chadd and Extence 2004) combines data on the average rarity of species in a sample, a weighted taxon richness (the BMWP score) and the rarest species in a sample. It reflects an aggregate conservation value of a macroinvertebrate sample. CCI has been applied to local investigations, but has not been applied at a national scale before, probably because there is a lack of national-scale datasets that include macroinvertebrate taxa identified to species level. CCI is expressed as an Observed score, not an Observed ÷ Expected ratio (see Appendix 2.1).

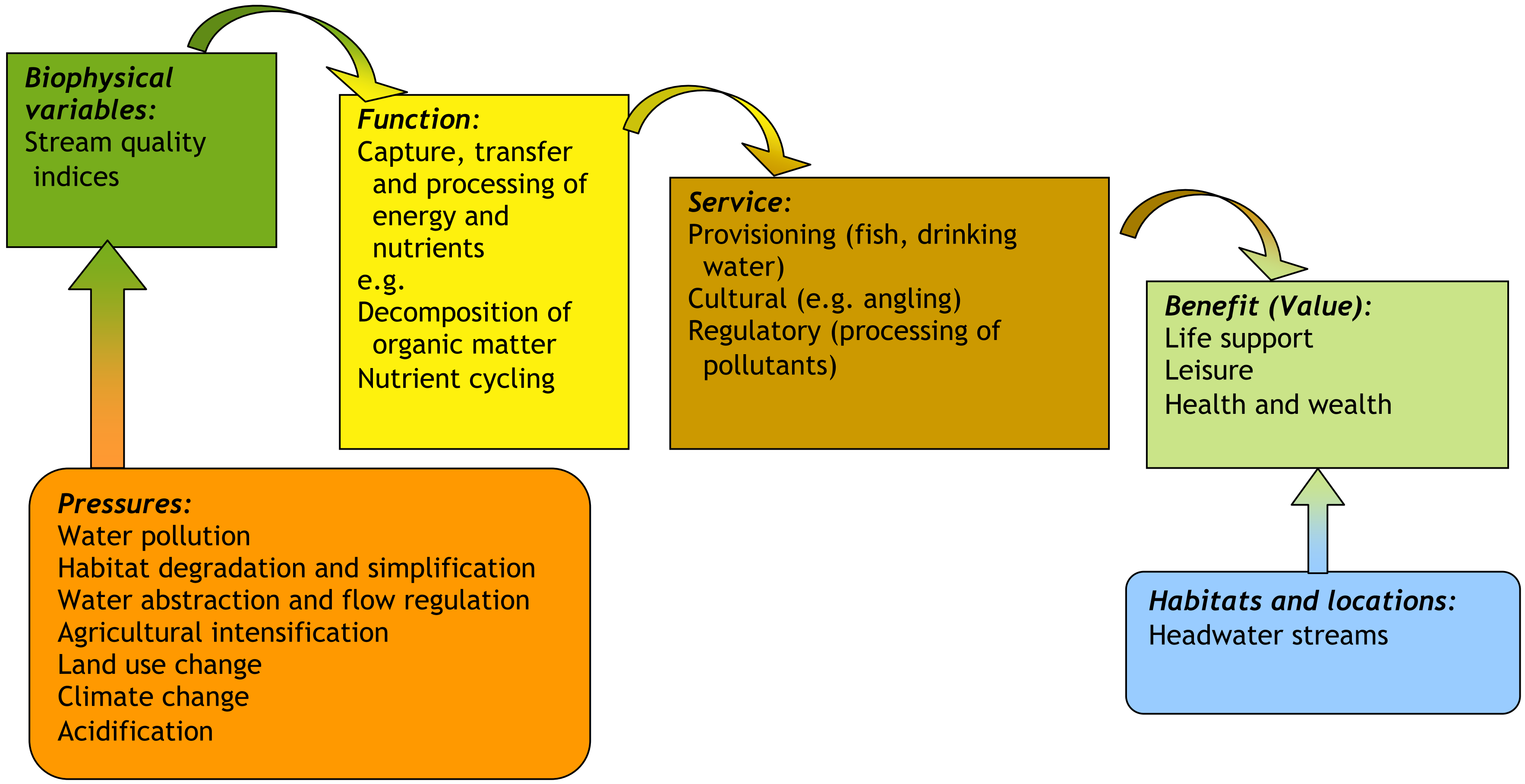
For each of these indices, a higher value for the index can be considered to be a better biological condition.

Biophysical measurement link to ecosystem service: clean water provision

Provision of sufficient quantities of clean water is an important ecosystem service delivered by headwater streams. Furthermore, headwater streams are important reservoirs of biodiversity (Furse *et al.* 1991; Meyer *et al.* 2007; Clarke *et al.* 2008). Environmental change, including climate change, is a major risk for clean water provision and biodiversity. Water quantity and quality show considerable natural variation, however freshwaters have been exploited by humans for thousands of years both for clean water supply and as a conduit for waste water disposal. This exploitation is increasingly associated with environmental degradation (Malmqvist and Rundle 2002; Nilsson *et al.* 2005).

Biological monitoring of water quality using macroinvertebrates has a long history (Hellawell 1978; Jones *et al.* 2010), further details are contained in the Appendix 2.1. Figure 2.1 illustrates the proposed ecosystem services cascade for clean water.

Figure 2.1: The ecosystem service cascade for clean water (after Haines-Young and Potschin 2007).



2.3 Current status and trends

Current status and trends since 1990 for macroinvertebrate Average Score per Taxon (ASPT) and number of Biological Monitoring Working Party macroinvertebrate scoring taxa (TAXA) are described in the Countryside Survey Headwater Streams Report for 2007 (Dunbar *et al.* 2010a). The patterns for the Observed indices, and Observed ÷ Expected sample values for these indices, are broadly similar. These results are summarised in the next two paragraphs.

Average Score per Taxon has shown a significant increase from 1990 to 1998 in England and Scotland. This trend has continued in England, albeit at a lower rate, from 1998 to 2007. For Scotland, there was no significant difference from 1998 to 2007. Observed changes in ASPT in Wales were not significant, though this may be due to the lower sample sizes for Wales: it may be that the apparent patterns in mean scores simply reflect noise, with no actual trend, or it may be that the apparent trends follow the true patterns, but that number of samples, and hence statistical power, is not sufficient to detect them. Improvements from 1990 to 1998 may partly be due to the extensive drought covering England and Wales in 1990, potentially lowering ASPT scores.

For number of Biological Monitoring Working Party scoring taxa, results for England mirror the results for ASPT; with continuing improvements throughout the survey period, although the increases from 1998-2007 appear relatively stronger for these measures than for ASPT. Both Wales and Scotland show significant increases from 1990-1998. Decreases occur from 1998 to 2007, however, only the results for Scotland are significant.

Status and trends for macroinvertebrate Community Conservation Index (CCI) are described here (Fig. 2.2), as they were not included in Dunbar *et al.* 2010a. Increases in CCI in Scotland from 1990 to both 1998 and 2007 were statistically significant, as were changes in England from 1990 to 2007. Results for Wales follow the pattern of those for Scotland, but, as in several of the headwater streams analyses, due to the lower sample sizes, it is not possible to tell whether the observed changes are random noise or true changes in mean scores.

In Countryside Survey, England and Scotland are each divided into three Environmental Zones, Wales is divided into two Zones (Carey *et al.* 2008). For Scotland and Wales, there was no evidence for differences in CCI between Environmental Zones. For England from 1998 to 2007, there was some evidence for a difference in trends between Environmental Zones (EZs), but slightly weaker evidence that trends for individual EZs differ significantly from zero. The pattern is for no change in EZ1, an increase in CCI in EZ2, and a decrease in CCI in EZ3.

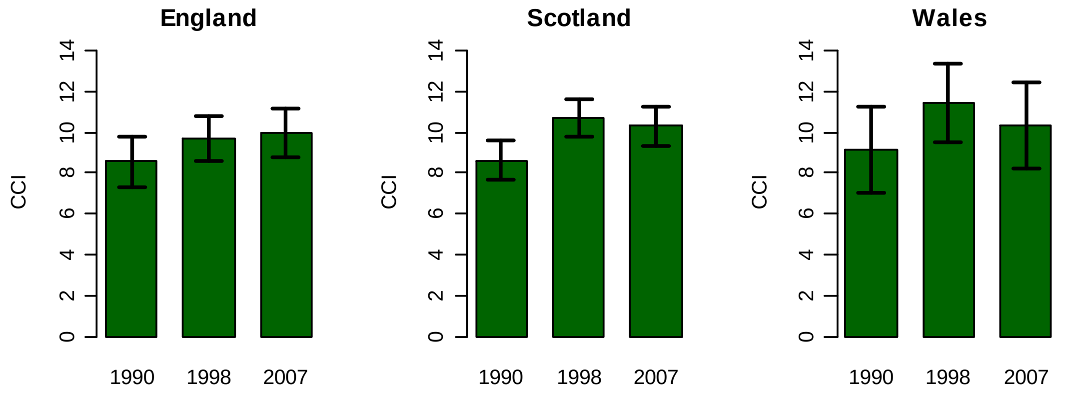


Figure 2.2: Mean values for Community Conservation Index (CCI) by country and year for all stream sites. Black bars represent 95% confidence intervals.

2.4 Attribution of spatial and temporal patterns

Explanatory variables

A number of Countryside Survey (CS) measures were considered as explanatory variables in the analysis. These measures were obtained from the River Habitat Surveys and chemistry samples taken at the macroinvertebrate sampling sites, or from the terrestrial surveys of each Countryside Survey square. The CS measures used represent different environmental characteristics which will themselves show responses to their environment. Hereafter, the term "CS measures" is used to refer to the measures used as explanatory variables in the analysis. Data were available for 1990, 1998 and 2007 except where noted below. Further details on the data collection protocols are contained in Maskell *et al.* 2008, Smart *et al.* 2008 and Murphy and Weatherby 2008.

The following CS measures were matched on a square / survey basis to the macroinvertebrate indices outlined above:

- **In-stream**
    - Soluble reactive phosphorus (SRP) concentration in mg/l (1998, 2007 only) taken from a stream water sample at the time of the headwater stream survey. Phosphorus is considered the most important cause of eutrophication in temperate rivers, as it is thought to be the limiting nutrient for algal growth. SRP is broadly equivalent to bioavailable phosphorus. SRP levels from a single spot sample can be dependent on antecedent streamflow conditions;

  - Extent of stream channel resectioning, as quantified by the River Habitat Survey (Raven *et al.* 1998a; Raven *et al.* 1998b) Habitat Modification Resectioned Bed and Banks Sub-Score (1998, 2007 only) undertaken as part of the headwater stream survey. Higher values indicate a greater extent of resectioning in the 500m River Habitat Survey reach;
  - Sub-scores from the River Habitat Survey Habitat Quality Assessment (1998, 2007 only): flow types, channel substrate, channel features, channel vegetation. Higher values indicate more diverse / higher quality conditions. Sub-scores for bank vegetation, land use, trees and associated features and special features were not used.
- **Along-stream**
  - Cover Weighted Canopy Height (averaged for all streamside vegetation plots in each square for each survey). This is calculated using a trait database comprising the potential heights of the plant species found in Countryside Survey, weighted by the observed cover values for each plant species in each plot. Canopy height is sensitive to a range of factors including succession, grazing, fencing, recent flood history, riparian mowing/channel maintenance and eutrophication. Previous work has shown that it is likely to reflect a eutrophication-related stimulation of competitive nutrient demanding taller herbs as well as a successional stimulus to biomass production (Smart *et al*. 2005);
  - Percentage cover of woody species (% Woody Cover), averaged for all streamside vegetation plots in the square. Compared to Cover Weighted Canopy Height, this is much more sensitive to the realisation of late-successional conditions and less sensitive to eutrophication;
- **Wider landscape**
  - Percentages of selected Broad Habitat types in the 1km Countryside Survey square containing the headwater streams site: Arable, Improved Grassland and Urban. The focus was on a few “intensive” categories to try to avoid problems with correlations between percentages of different categories which in total add to 100%;

In the evaluation of the relationship between certain chemical characteristics, the following additional environmental characteristics were used:

- In-stream: total oxidised nitrogen (TON) concentration (2007 only) taken from a stream water sample at the time of the headwater stream survey;
- Wider-landscape: soil Olsen Phosphorus, in mg/kg (1998, 2007), representing phosphate in the soil available to plants. The mean of all available soil samples in the square was taken (maximum of five taken from large plots within major Broad Habitats: Emmett *et al.* 2008).

Biotic scores (Response variables)

The number of macroinvertebrate samples for which values for O/E (observed ÷ expected) Average Score per Taxon and number of Biological Monitoring Working Party taxa were available are shown in Table. Values for Community Conservation Index were also calculated for this same set of samples. These were matched to the explanatory variables on a square and survey year basis.

Analyses undertaken

As noted above, two subsets of this dataset were created (see Table 2.1): one included all of the above explanatory variables and biotic scores for 1998 and 2007 only (the “short” dataset). The other included the above variables which were available for 1990, 1998 and 2007 (the “long” dataset). Further technical details of the data processing and statistical methods are contained in the Appendix 2.1.

Mixed effects multiple linear regression models were built with each of the two subsets for Average Score per Taxon, number of Biological Monitoring Working Party scoring taxa and Community Conservation Index, making six different models in total.

In order to visualise the relative strengths of effects of explanatory variables in explaining spatial pattern of selected response variables, hierarchical partitioning models were built using the explanatory variables identified in the mixed modelling process. These were built using the variables for each square, averaged for 1998 and 2007; hence they illustrate spatial but not temporal patterns.

Correlations between selected variables were examined by visual inspection of scatter plots. Spatial correlations were examined using the square by square data averaged for 1998 and 2007. Relationships between selected correlated CS measures used in the modelling of biotic indices were investigated using the same mixed model regression approach outlined above, using the following sets of variables:

- Soluble Reactive Phosphorus and River Habitat Survey Resectioned Bed and Banks Sub-Score as response variables for the 1998 and 2007 data; explanatory variables used in the biotic scores analysis were used (except the variable used as the response);
- Soluble Reactive Phosphorus as a response variable and soil Olsen Phosphorus as an explanatory variable for the 1998 and 2007 data;
- Soluble Reactive Phosphorus as a response variable and Total Oxidised Nitrogen, Broad Habitat % cover values as explanatory variables for the 2007 data.

Selected temporal correlations were examined by examining the full 1998 and 2007 dataset, but with the square-by-square average of each variable subtracted.

Neither Soil Olsen Phosphorus nor Total Oxidised Nitrogen was used in the models with the biotic scores as response variables.

Table 2.1: Numbers of sites and samples used in different analyses

Analysis	Number of sites	Number of samples
1990-1998-2007	249	699 (697 for CCI)
1998-2007	241	463
1998-2007 (Soil Olsen Phosphorus)	172	344

2.5 Results

Results are presented for analyses of the 'long' dataset (including data from 1990, 1998 and 2007) and 'short' dataset (1998 and 2007 only).

Relationships between the biophysical explanatory variables are described first. Secondly, the relationships between the stream biotic indices (used as ecosystem service measures) and the biophysical explanatory variables are described.

Relationships are described as positive or negative. A positive relationship means that the response variable changes in the same direction as the explanatory variable, i.e. an increase in the explanatory variable is associated with an increase in the biotic score, and that a decrease in the explanatory variable is associated with a decrease in the biotic score. A negative relationship means that the response variable changes in the opposite direction to the explanatory variable, i.e. an increase in the explanatory variable is associated with a decrease in the biotic score (and vice-versa) (Table 2.2).

Table 2.2: Summary description of term 'positive relationship' and 'negative relationship'

Relationship	Explanatory variable	Response variable
Positive	↑	↑
Positive	↓	↓
Negative	↑	↓
Negative	↓	↑

Correlations between individual explanatory variables

There were some clear correlations between explanatory variables (Fig. 2.3). Arable, Intensive Grassland and Urban Broad Habitat percentages are inevitably correlated as they represent proportions of a set number of categories. There was also a clear positive correlation between Streamside Plot % Woody Cover and Cover Weighted Canopy Height. This is logical as they are based on the same underlying vegetation data. It was felt important to keep both variables in the analysis as there is still considerable scatter in the relationship and they do quantify different aspects of streamside vegetation. There were positive correlations between Cover Weighted Canopy Height and % Arable and % Improved Grassland.

Correlations between other variables were sometimes less obvious, e.g. the relationship between Soluble Reactive Phosphorus concentrations and River Habitat Survey Resectioned Bed and Banks Sub-Score became apparent as the modelling progressed. This illustrates that care should be taken in interpreting apparent correlation (or lack of correlation) between pairs of variables in the two-way scatter plots: in reality these plots may hide more complex underlying relationships (see Box 1.2).

Furthermore, there were some correlations between the changes through time in the Countryside Survey squares. Most noticeably, between 1998 and 2007, change in Broad Habitat % Arable and % Improved Grassland were correlated (Fig. 2.4).

Mixed-effects multiple regression models

Table 2.2 2.3 summarises the retained explanatory variables in each of the six models built to link CS measures to biotic indices representing headwater stream biological water quality and biodiversity. In addition, two models examining relationships between explanatory variables are also reported.

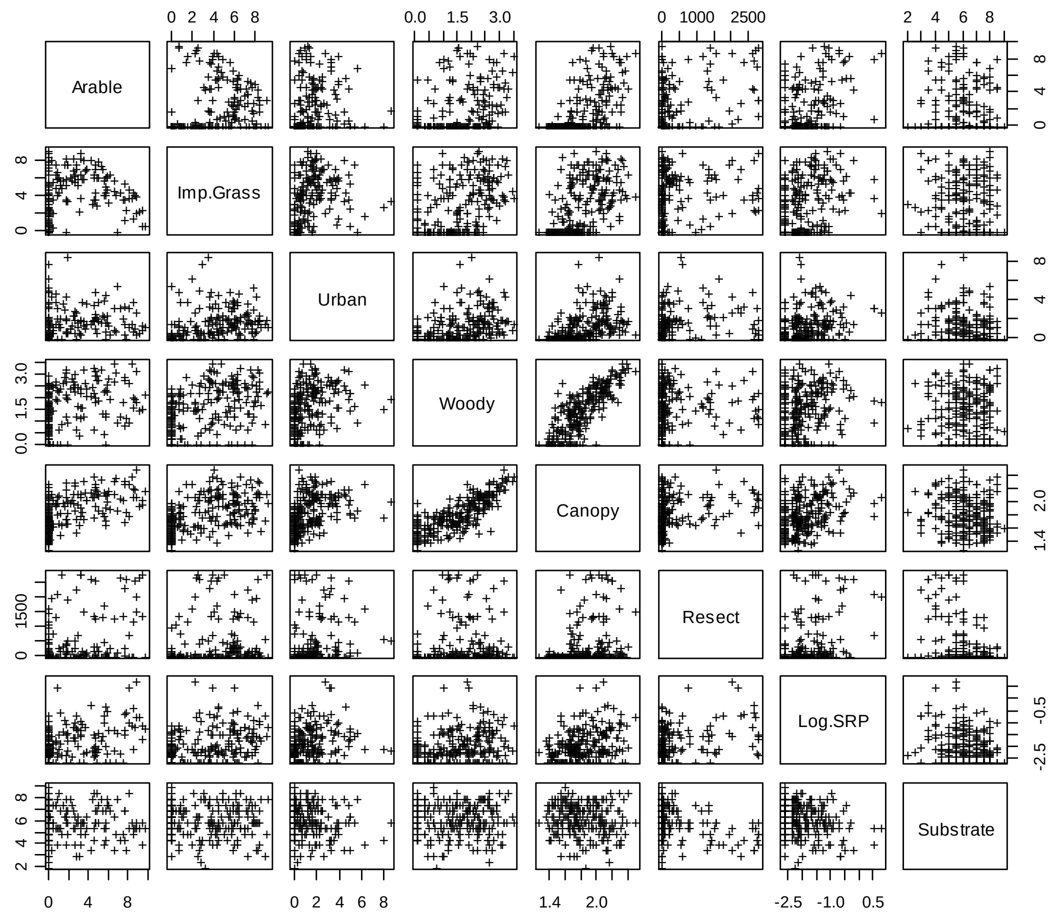


Figure 2.3: Scatter plot between major explanatory variables. Broad Habitat (% Arable, % Improved Grassland (Imp.Grass) and % Urban of 1km CS square) and Streamside Plot % Woody Cover (Woody) variables are square root transformed. Canopy refers to Streamside Plot Cover Weighted Canopy Height, Resect (Resectioned Bed and Banks) and Substrate (Channel Substrate) refer to River Habitat Survey Sub-Scores. Log.SRP refers to Soluble Reactive Phosphorus concentration (µg/l) log(x+0.0025) transformed. Data shown from 1998 and 2007 only.

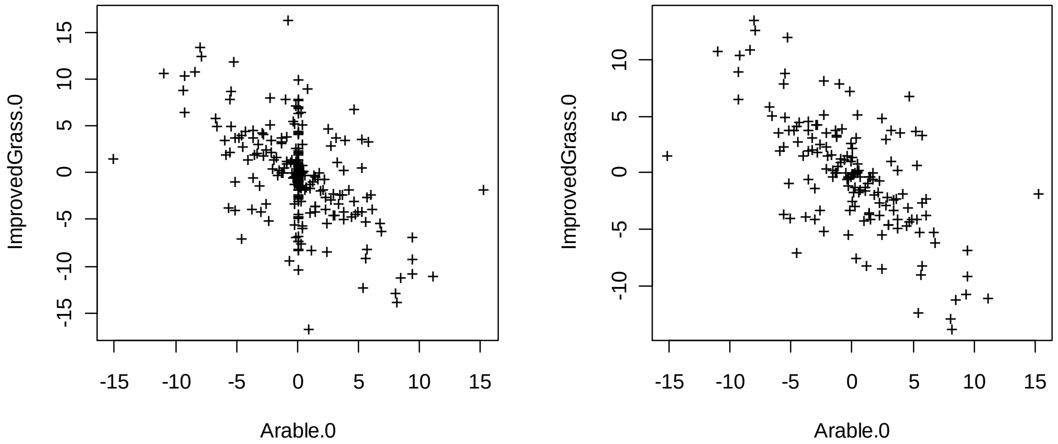


Figure 2.4: Relationship between change in Countryside Survey square level Broad Habitat % Arable from 1998 to 2007 and change in % Improved Grassland, illustrating the close relationship. Left plot includes all sites, right plot excludes those with mean % Arable <1. Variables are centred by square, hence zero indicates no change.

Table 2.3: Summary of relationships between Countryside Survey measures and headwater stream macroinvertebrate scores and associated variables.

	Explantory variables					
	1990, 1998, 2007 (long dataset)			1998-2007 (short dataset)		
Response	**Type**	**Variable**	**Direction**	**Type**	**Variable**	**Direction**
Average Score per Taxon (biological water quality)	BH	% Arable	-	BH	% Arable	-
	BH	% Improved Grassland	-	BH	% Improved Grassland (Spatial and Temporal*****)	-
	BH	% Urban	-	SSP	% Woody Cover	+
	SSP	Woody Cover (Spatial and Temporal*****)	+	HS	Soluble Reactive Phosphorus concentration	-
	SSP	Cover Weighted Canopy Height	-	HS	RHS channel substrate richness	+
Community Conservation Index (biodiversity)	BH	% Arable	-	HS	RHS resectioned Bed and Banks Sub-Score	-
	BH	% Urban	-	HS	Soluble Reactive Phosphorus concentration	-
	SSP	% Woody Cover	+	SSP	Cover weighted canopy height (marginal)	+
BMWP taxon richness (water quality and biodiversity)	BH	% Improved Grassland	+	BH	% Improved Grassland	+
	SSP	% Woody Cover	+	HS	Soluble Reactive Phosphorus concentration	-
	SSP	Cover Weighted Canopy Height	-	HS	RHS Resectioned Bed and Banks Sub-Score	-
Soluble Reactive Phosphorus (SRP)				BH	% Arable	+
				BH	% Improved Grassland	+
				BH	% Urban	+
				HS	RHS Resectioned Bed and Banks Sub-score	+
River Habitat Survey Resectioned Bed and Banks Sub-Score				BH	% Arable	+
				BH	% Urban	+
				HS	Soluble Reactive Phosphorus concentration	+

- Direction: + = positive relationship between explanatory and response variable, - = negative relationship between explanatory and response variable (see
- *Table 2.2*)
- Data type: BH = Broad Habitats data, SSP = Streamside Plots data, HS = headwater streams data
- All effects spatial (between square) only unless otherwise noted (*****)
- The parameter effects are from multiple regression models, i.e. for a particular model, the effects are to be interpreted with the other parameters held constant.

Macroinvertebrate Average Score per Taxon (ASPT)

There is a consistent negative between-square (i.e. spatial) relation between macroinvertebrate Average Score per Taxon (ASPT) and area of intensively managed Broad Habitat, particularly % Arable and % Improved Grassland. There is a positive relationship between mean % Woody Cover in the Streamside Plots and ASPT, notably this is both a between-square (spatial) and a temporal effect for the longer (1990-2007) dataset.

The fact that the temporal component of Streamside Plot % Woody Cover is not statistically significant in the analysis of the shorter (1998-2007) dataset is almost certainly a statistical power issue, losing the 1990 data gives less variation in time-varying % Woody Cover and fewer data points.

Moving from the longer dataset to the shorter dataset reduces the data available, but allows the use of additional explanatory variables: River Habitat Survey Sub-Scores and Soluble Reactive Phosphorus concentration. In virtually all cases, when comparing the analyses of the longer and shorter datasets the variables that drop out in terms of statistical significance for the shorter dataset have comparable parameter values in both analyses, they simply lose precision (and hence significance at p=0.05) when the 1990 data are removed. However, in other cases the Broad Habitat cover types become superfluous because other variables dominate. In particular, % Urban would be removed because Soluble Reactive Phosphorus concentration has a more precisely defined relationship with Average Score per Taxon. Furthermore, removing % Urban helps improves significance of % Arable in this case, suggesting a correlation between these two contrasting stressors.

For the shorter dataset, there was a negative temporal relationship between Broad Habitat % Improved Grassland and Average Score per Taxon (ASPT): i.e. squares where the cover of Improved Grassland had increased over time were associated with lower ASPT scores, and vice-versa. This effect is not apparent in the larger dataset.

When considered on its own, the River Habitat Survey Resectioned Bed and Bank Sub-Score has a negative relationship with Average Score per Taxon, however considering the wider range of explanatory variables, this association is over-ridden in the model by the effect of Soluble Reactive Phosphorus concentration on Average Score per Taxon. This suggests a correlation between these two variables, and potentially others too.

Because of the potential for shading to influence macrophyte, bryophyte and algal composition and production, and hence have secondary impacts on the macroinvertebrate community, both Streamside Plot % Woody Cover and Cover Weighted Canopy Height were tested for an interaction effect with Soluble Reactive Phosphorus concentration. There was weak evidence (not statistically significant at p=0.05) for an interaction between Cover Weighted Canopy Height and Soluble Reactive Phosphorus affecting Average Score per Taxon. This would have the effect that where the riparian canopy was high, the effects of Soluble Reactive Phosphorus on ASPT are lessened (the

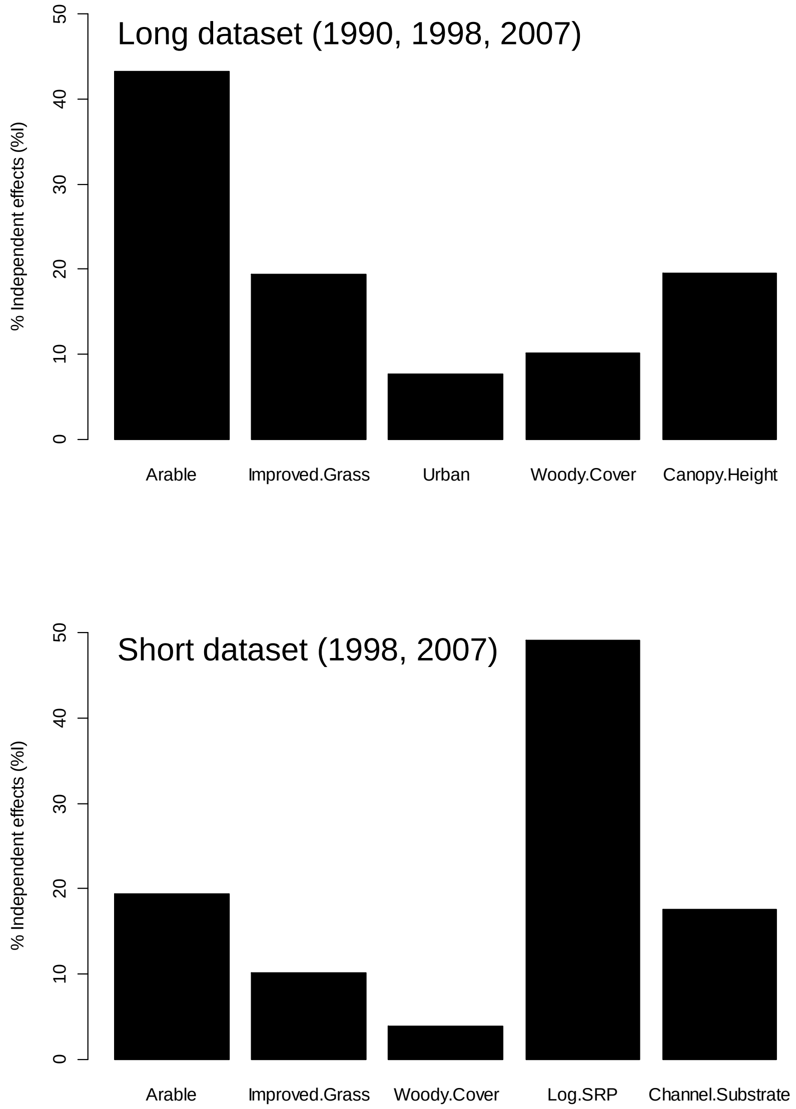


Figure 2.5: Relative magnitude of independent effects on spatial pattern of Average Score per Taxon in long (top graph) and short (bottom graph) datasets. Arable, Improved.Grass and Urban are % Broad Habitat cover values in 1km CS square. Woody Cover is % Woody Cover and Canopy.Height is Cover-Weighted Canopy Height, both in Streamside Plots, averaged within each CS square. Log.SRP is the log concentration of Soluble Reactive Phosphorus in the spot water sample, Channel.Substrate is the River Habitat Survey Channel Substrate Sub-Score.

expected direction of the effect). % Woody Cover appeared not to have this effect.

The relative strengths of the individual explanatory variables in explaining the spatial pattern of Average Score per Taxon, for both the large and small datasets are illustrated in Figure 2.5. The signs of the effects are as denoted in Table 2.3 above, i.e. all negative except Streamside Plot % Woody Cover and River Habitat Survey Channel Substrate Sub-Score.

Macroinvertebrate Community Conservation Index (CCI)

For the long dataset, the important explanatory variables for Community Conservation Index are a subset of those seen with Average Score per Taxon, i.e. Broad Habitat % Arable, Broad Habitat % Urban and Streamside Plot % Woody Cover. The result for the short dataset is a contrast with that of the long: when the additional variables of Resectioned Bed and Banks Sub-Score and Soluble Reactive Phosphorus are introduced, the higher precision of their effects decreases the precision of the partial relationships with the square-level Broad Habitat data, the latter are consequently dropped from the model. When the significant variables for the long dataset are used on their own (i.e. without the additional variables available for the short dataset) as explanatory variables with the short dataset they are still all significant and comparable in effect size.

A comparison of the short dataset results for Average Score per Taxon (ASPT) and Community Conservation Index (CCI) is interesting. It is again worth noting that Resectioned Bed and Banks Sub-Score is a significant predictor in the model for Average Score per Taxon, providing Soluble Reactive Phosphorus concentration is not also in the model. However, for the Community Conservation Index model, it is Resectioned Bed and Banks Sub-Score and Soluble Reactive Phosphorus concentration that both have negative effects, despite the fact that they are partially correlated. Several factors may be influencing the difference in the response behaviour of CCI and ASPT. Firstly, there are a number of species which are relatively insensitive to poor water quality (particularly organic pollution), but which are relatively rare, so score highly in CCI. These include a number of beetles and diptera (fly) larvae. Secondly, CCI is calculated from species data, whereas ASPT is calculated from sample data aggregated to family taxonomic level. Thirdly, there are entire families whose species contribute to CCI but which are not included in the calculation of ASPT.

Number of Biological Monitoring Working Party scoring macroinvertebrate taxa

Number of scoring taxa has traditionally been used as part of the national scheme for biological water quality assessment. It is a measure of richness of macroinvertebrate families (i.e. not species) and only includes families that are assigned a score under the BMWP system. It would be expected to respond to a range of stressors, including toxic chemicals such as pesticides and metals. As with Community Conservation Index, there are differences in the optimum set of explanatory variables for the long and short datasets. With the latter dataset, it responds negatively to Soluble Reactive Phosphorus concentration and River Habitat Survey Resectioned Bed and Banks Sub-Score. This is notable as it might be thought that in naturally nutrient-poor streams, low to moderate increases in nutrients such as Soluble Reactive Phosphorus are generally positively associated with taxon richness.

For the long dataset, as for Average Score per Taxon (ASPT), there are positive effects of % Woody Cover, and negative effects of Cover Weighted Canopy Height mirror the result found for ASPT. However, there is a positive effect of % Improved Grassland, this contrasts with the results for ASPT with which % Improved Grassland had a negative relationship. It is not immediately obvious why this might be. It may be related to other covariates: a similar but less pronounced effect exists in an apparent positive relationship between time varying Broad Habitat % Arable and ASPT which appears to relate to reversion from Arable to Improved Grassland from 1990 to 2007.

Soluble Reactive Phosphorus concentration and River Habitat Survey Resectioned Bed and Banks Sub-Score

Given the behaviour of these two variables in the models, it was felt to be valuable to examine how they relate to each other, and to the Broad Habitat variables. Soluble Reactive Phosphorus concentration was positively related to % Arable, % Improved Grassland, % Urban and River Habitat Survey Resectioned Bed and Banks Sub-Score together.

Nutrients in soil and water

Using data from 1998 and 2007, there was no significant relationship between mean (all samples in square) soil Olsen Phosphorus concentration and Soluble Reactive Phosphorus concentration in the stream water.

Using 2007 data, there was a highly significant relationship between Soluble Reactive Phosphorus (SRP) and Total Oxidised Nitrogen concentration in the stream water (Fig. 2.6), with a correlation coefficient of almost 1; however there are also relationships between the other plotted variables, particularly between Broad Habitat % Arable and the nutrients. Hierarchical partitioning, using Soluble Reactive Phosphorus concentration as the response and Broad Habitat % Arable and % Improved Grassland as explanatory variables, demonstrated that 87% of the variance in Soluble Reactive Phosphorus concentration was explainable by Total Oxidised Nitrogen concentration, 9.3% by Broad Habitat % Arable and 3.7% by % Improved Grassland. Six samples were clear outliers in this relationship, having much lower SRP concentrations than their land cover and SRP concentrations would suggest from a linear relationship.

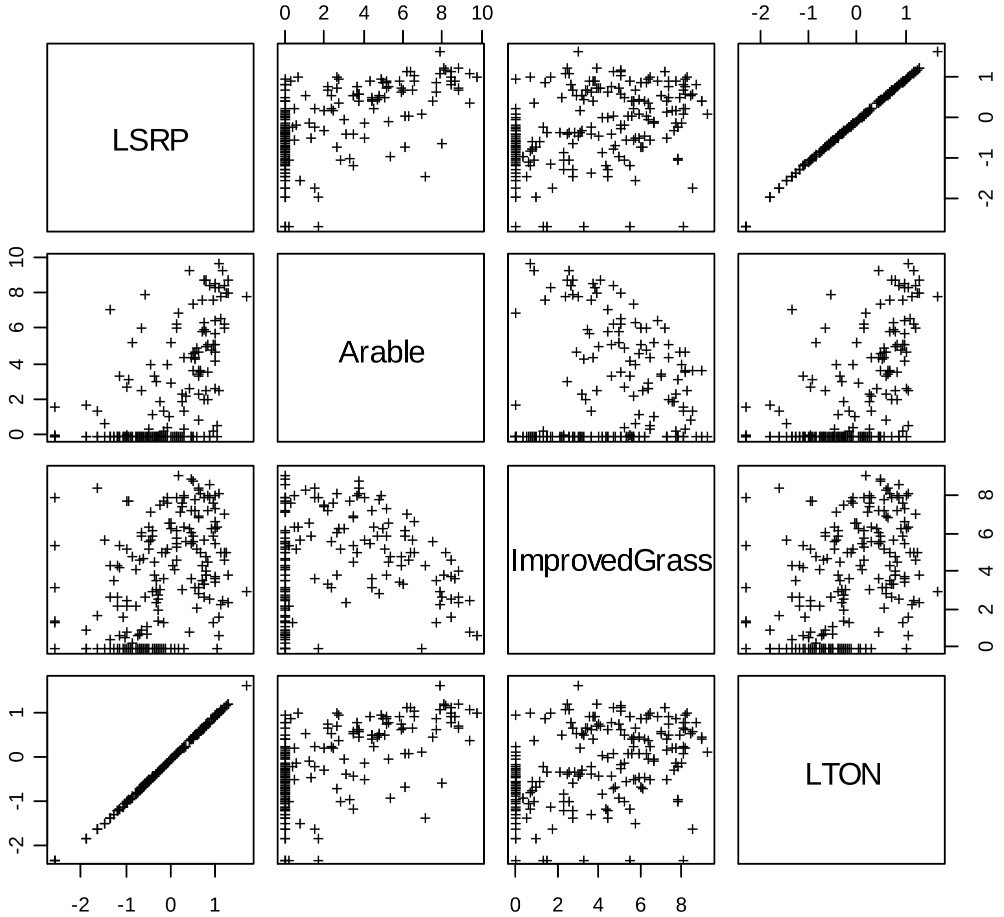


Figure 2.6: Relationships between stream Soluble Reactive Phosphorus concentration (LSRP), Total Oxidised Nitrogen concentration (LTON) (both log transformed) and cover of intensive Broad Habitat types for 2007 data. % Arable and % Improved Grassland are square root transformed. Note that there is some over-plotting of points, particularly corresponding to the lowest nutrient concentration points, which correspond to the below detection limit value.

2.6 Discussion

Context

Intensive land management activities can have a negative effect on stream water quality (Baker 2003). Further, many studies have demonstrated links between catchment land use, particularly the quality of the riparian corridor and stream ecological status (Strayer *et al.* 2003; Allan 2004). However, such studies are often based either exclusively on spatial rather than temporal data, or on temporal data from relatively few locations. The former also often rely on remotely-sensed data on land usage to act as a surrogate for the actual, more proximate, factors which directly affect stream biota. Much of this knowledge is derived from studies in North America, Australia and New Zealand, although a few studies have been undertaken in Europe. Few studies have examined both spatial and temporal patterns and few studies quantify effects measured at multiple spatial scales, i.e. wider landscape (e.g. land uses) riparian/along stream (e.g. woody cover) and instream (e.g. stream water quality, physical habitat) factors together. Hence the results from this study are genuinely novel and of considerable scientific interest.

General observations

Broadly, the patterns observed in this analysis are associated with differences in the environmental characteristics of squares (spatial patterns), rather than in the changes occurring in the squares between surveys (temporal patterns). There could be several reasons for this:

- The observed temporal changes in environmental characteristics (both Broad Habitat and Streamside Plot) on a square by square basis may not be of great enough magnitude to change in-stream communities;
- There may be temporal relationships between these variables, but Countryside Survey does not yet have enough temporal data (hence statistical power) to define them;
- There may be a temporal lag in the biological response to the observed environmental changes;
- There may be relationships acting over timescales shorter than the inter-survey period;
- There may be temporal relationships, but the correct variables have not been used;
- The observed temporal variation in the biota may be more related to internal stream processes and/or “noise”.

Given the strong spatial patterns in the dataset, and the limited number of time points, it is notable that in some cases, temporal associations have been detected, both with a more locally-acting variable (% Woody Cover) and a more indirectly acting variable (Broad Habitat % Improved Grassland).

This study did not explicitly consider the fact that certain CS measures, such as those of the Streamside Plots, are both response variables and explanatory variables. This would be possible using a range of path analysis techniques such as structural equation modelling.

In this study, it was decided to use cover of selected Broad Habitats in survey squares (See Appendix 2.1 for more information on this decision). It is interesting that relationships were found with cover of intensively managed Broad Habitat types in the square, despite the fact that some of the catchment for each headwater stream site will be outside the square, and some of the square will be outside the catchment. There are at least two potential reasons for this apparent contradiction:

- It may be that this reflects the proximity of the CS measures and the stream site, i.e. impacts of intensive management may become less the further they are located from the site of interest; or
- It may simply be that land cover in the Countryside Survey square is similar to the land cover in the entire catchment of the headwater stream site.

In practice the relationships between CS measures means that in some cases, although there may be individual univariate relationships between a biotic score and several measures, together, only a subset of those measures are needed. For example, either Soluble Reactive Phosphorus concentration or extent of Resectioned Bed and Banks, but not both together influences Average Score per Taxon. However in other cases, some of these inter-correlated variables have significant explanatory power together, giving an additive effect (e.g. both Soluble Reactive Phosphorus concentration and extent of Resectioned Bed and Banks influence Community Conservation Index and number of BMWP scoring taxa).

In the following sections, as elsewhere in the report, references to macroinvertebrate Average Score per Taxon and number of Biological Monitoring Working Party scoring macroinvertebrate taxa refer to their Observed ÷ Expected (O/E) ratios, and not their raw values (see Appendix 2.1 for further details). However, references to macroinvertebrate Community Conservation Index (CCI) refer to the raw scores.

Broad Habitat % Arable in square

The negative association between % arable land and various biotic scores is one of the least surprising results as studies in other countries have demonstrated similar results. As with all the Broad Habitat and Streamside Plot variables, there are multiple potential mechanisms by which an increasing % cover of arable land near a stream can have a negative effect on the stream macroinvertebrate community. It is notable that Broad Habitat % Arable, together with other potentially correlated variables such as % Improved Grassland and Soluble Reactive Phosphorus concentration have a combined, additive and negative effect on macroinvertebrate Average Score per Taxon.

% Woody Cover in Streamside Plots

The changes (generally increases) in cover of streamside woody vegetation have already been noted as being one of the major changes observed since 1990 (Carey *et al.* 2008). Increases in streamside woody cover have been shown to be associated with decreases in streamside plant species diversity, nectar plant diversity and appropriate diversity (Carey *et al.* 2008 and Chapter 5 of this report).

In this study, Streamside Plot % Woody Cover shows both a positive spatial and a positive temporal relationship with macroinvertebrate Average Score per Taxon. There are several potential reasons for this, including:

- Woody cover near streams provides habitat for the adult aerial life stages of insects with aquatic larval stages;
- Woody cover near streams should be associated with in-stream wood and dead leaves both of which provide hydraulic diversity and habitat for many different taxa and food for macroinvertebrates (e.g. some caddis fly species eat wood, many taxa including some caddis fly species, shrimp and hoglouse eat leaves);
- Shading from woody cover will affect stream temperatures and the amount of light reaching the stream bed. Both of these factors will influence in-stream macrophyte and benthic algal composition and coverage;
- Areas with streamside woody cover may act as buffer strips which may trap and buffer nutrients flowing from land to water. However, the effects of buffer strips are still controversial.

Cover Weighted Canopy Height of Streamside Plots

The negative spatial relationship between macroinvertebrate Average Score per Taxon and Cover Weighted Canopy Height is notable, as it is apparent despite the strong relationship with Streamside Plot % Woody Cover. The relationship may be because this variable is in part a surrogate for nutrients in the streamside soil. This is a complex issue, as there are multiple competing mechanisms. Tall herbs could also be expected to provide habitat for adult insects, inputs of detritus and shading for the stream itself.

The possibility of riparian vegetation influencing in-stream primary production via shading effects was not strongly supported, but the effect was in the right direction, and could be investigated further with more targeted analysis.

River Habitat Survey Channel Substrate Sub-Score

The effects of Channel Substrate Sub-Score (effectively a richness of dominant substrate types) is interesting. It is well known that substrate plays an important role in the distribution of macroinvertebrates (Erman and Erman 1984; Beisel *et al.* 1998; Heino *et al.* 2003). It is also worth noting that in Countryside Survey, the macroinvertebrate sampling site will be of the order of metres, to a few tens of metres, of river length (longer in narrower streams), whereas the associated River Habitat Survey is carried out over a 500m reach around the macroinvertebrate sampling site. Substrate at the macroinvertebrate sampling site is quantified at the time of sampling as % cover of various substrate size classes, it is then used (as a single mean *phi* value) as one of the predictor variables used to calculate an expected value for biotic indices such as Average Score per Taxon. The question is why is there then an additional effect of substrate richness at a larger spatial scale? It could be that richness of substrate sizes, a surrogate for habitat complexity is associated with higher ASPT scores, further it is entirely possible that substrate richness over a wider spatial scale than the RIVPACS sampling reach is important for more local biological quality (e.g. see Kail and Hering 2009). This could be via several mechanisms such as provision of habitat for colonists to other areas and retention of detritus.

Broad Habitat % Improved Grassland in square

The negative spatial and temporal relationships between Broad Habitat % Improved Grassland and Average Score per Taxon (ASPT) are interesting. This means firstly that spatially, squares with more % Improved Grassland have lower ASPT scores, and vice-versa, controlling for the other variables in the model (i.e. when they are held constant). Secondly, squares where % Improved Grassland increased from 1998 to 2007 had decreased ASPT scores (again controlling for other variables), and vice-versa. Between 1998 and 2007 there was a 5.4% increase in Improved Grassland area across Britain resulting largely from gains at the expense of Arable and Neutral Grassland (Carey *et al.* 2008). The temporal relationship may be reflecting this reversion. Intensification from Neutral to Improved Grassland might be expected to be associated with lower stream biological quality. However, reversion from Arable to Improved Grassland might initially be expected to be beneficial to stream biological water quality. The results imply that this may not be the case: this reversion can be detrimental to stream biota. It may be that the negative impact of intensification from Neutral Grassland outweigh any benefits of reversion from Arable to Improved Grassland. This highlights the need to examine the nature of temporal changes in land use, not simply changes in overall proportions. Alternatively, the observed trends for arable areas to be farmed less intensively may be outweighed by trends for Improved Grassland to be farmed more intensively. It would be premature at this stage to make firm conclusions on the mechanisms for the negative effects of Improved Grassland. They may relate to a negative association with increasing livestock density, which could be related to increased runoff of nutrients or toxic organic chemicals or poaching (trampling) by livestock of riparian areas.

River Habitat Survey Resectioned Bed and Banks Sub-Score

Between Countryside Survey squares, there was a negative relationship between extent of resectioning, as quantified by the River Habitat Survey and all three biotic scores examined. However, in the case of Average Score per Taxon (ASPT), this may be due to the spatial correlation of extent of resectioning with elevated nutrient levels. Although in this instance, Soluble Reactive Phosphorus concentration is a more dominant variable than extent of resectioning, it is possible that the resectioning also has an independent effect on ASPT, but that across the Countryside Survey dataset, there is not enough independence between these two variables for this to be apparent. Alternatively given the origins of Average Score per Taxon as a bioindicator of organic pollution, calculated from data aggregated to family level and with no weighting according to abundance, it might not be sufficiently sensitive. The LIFE index, originally designed to respond to hydrological change (Extence *et al.* 1999) would almost certainly be more sensitive to the effects of resectioning (Dunbar *et al.* 2010b; Dunbar *et al.* 2010c).

It is notable that for two out of three of the biotic indices examined (Community Conservation Index and number of BMWP scoring taxa), the between-square negative association with extent of resectioning held true whilst controlling for the large differences in headwater stream Soluble Reactive Phosphorus concentrations between squares. This is an important result, which confirms the negative association between historical and ongoing channel management activities and headwater stream biota.

Soluble Reactive Phosphorus concentration in water

There was a highly significant negative between-square (i.e. spatial) association between Soluble Reactive Phosphorus (SRP) concentration and all three indices examined, in some cases in addition to the effects of intensive land uses and other variables. In the modelling undertaken here, SRP concentration may well be acting partly as a surrogate for other nutrients, particularly nitrogen, but possibly also biochemical oxygen demand (BOD). Phosphorus in rivers is receiving considerable ongoing attention (Mainstone and Parr 2002; Hilton *et al.* 2006), however few studies have examined the effects of inorganic nutrients on headwater stream macroinvertebrates at broad spatial scales (e.g. Wang *et al.* 2007; Friberg *et al.* 2009). As well as natural background levels of inorganic nutrients, anthropogenic sources of inorganic nutrients will include effluent from sewage treatment works, septic tanks and agriculture (arable and livestock). Macroinvertebrates are not generally considered to respond directly to inorganic nutrients (in contrast to their well known responses to organic nutrients). However, inorganic nutrients such as SRP will have several indirect effects, influencing both macrophytes and benthic algal composition and biomass, which in turn provide habitat and food for macroinvertebrates. In addition, if nutrients were to cause excessive macrophyte or algal growth, this would deplete levels of dissolved oxygen in the water column at night, which would have major effects on the macroinvertebrate community.

The between-square (spatial) relationship between the extent of resectioning and Soluble Reactive Phosphorus concentration is not surprising, but there could be multiple mechanisms acting:

- It could be that both variables are responding to something else, e.g. they are a general consequence of the intensity of land use/ management in certain areas – i.e. they tend to go together. This could apply in both urban and rural settings (although Countryside Surveys squares are not heavily urban);

- There could be a more mechanistic explanation:
    - In rural areas, resectioning could indicate an absence of any buffer strips, and be more associated with under-drained land, indicating more direct connection between field and river. A similar argument would apply in urban areas;
    - In agricultural areas, lack of buffer strips and straightened river channels can enable accidental direct application of fertiliser to the stream.

Although it is Soluble Reactive Phosphorus (SRP) from the water column that is measured in Countryside Survey, an additional potential mechanism is that resectioned river stretches will have altered siltation patterns compared to more natural river stretches, which could lead to greater rates of sequestering of SRP from the water column by bed sediments (House and Denison 1997; Bowes and House 2001).

Interactions between the effects of Soluble Reactive Phosphorus, extent of channel degradation, shading of streams by riparian vegetation and hydrological regime strongly deserve further analysis. For example, there was weak evidence (not statistically significant at $p=0.05$) for an interaction between Cover Weighted Canopy Height and Soluble Reactive Phosphorus affecting ASPT. This would have the effect that where riparian canopy height was high, the effects of Soluble Reactive Phosphorus on Average Score per Taxon are lessened (the expected direction of the effect).

There are several potential reasons for the lack of a relationship between soil Olsen Phosphorus concentration and stream Soluble Reactive Phosphorus concentration for the 1998 and 2007 data. In particular, no attempt was made to select soil core locations which were most hydrologically linked to the stream. Furthermore, spot samples for nutrient concentrations will be influenced by antecedent weather conditions and flows.

The very strong relationship between Soluble Reactive Phosphorus (SRP) concentration and Total Oxidised Nitrogen concentration (TON) from the 2007 data is partly a function of the scale of Countryside Survey, there being samples at below detection limit for both determinands, and samples with high levels for both determinands. The hierarchical partitioning analysis demonstrates that despite the very high apparent correlation between SRP and TON, 13% of the variance in SRP is associated with the extent of cover of two square-level Broad Habitats (% Arable and % Improved Grassland). Overall, these results highlight the care that is needed in interpreting the results of the modelling of the impacts on the macroinvertebrate community: there are correlations between measured stressors, and by implication, other unmeasured stressors as well.

2.8 Conclusions

Despite the well-known associations between land management and stream ecological quality, the results from this study are genuinely novel and of considerable scientific interest. In this study, using data from across Great Britain, the negative effects of intensive land management on headwater stream biological water quality and biodiversity are broadly validated. No other study has looked at both temporal (multi-year) and spatial (hundreds of sites ranging over hundreds of km) patterns of stressors and stream ecological responses in this way. The study has confirmed that effects acting over multiple spatial scales can influence stream biota. Positive as well as negative effects have been demonstrated. In some instances more direct effects, such as spot measures of stream water quality, physical properties of the channel and of the riparian zone (such as amount of woody cover, diversity of streambed substrate sizes) seem to show the strongest relationships with stream biological water quality. In other instances, the strongest associations are with the extent of intensive land uses in the Countryside Survey square containing the headwater stream site. In this case, these extents act as useful surrogates for other unmeasured mechanisms of impact, such as sediment delivery, alteration of hydrological regime and pollution by organic chemicals.

Chapter 2: Online appendices

2.1: Further details on the derivation of biotic indices and statistical approaches to data analysis.

Chapter 3: Topsoil carbon

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Summary

- The soil component of Countryside Survey is unique as topsoil carbon concentrations (0-15cm) have been measured at three time points (1978, 1998 and 2007) together with topsoil bulk density (2007), a range of other soil parameters (1978, 1998 and 2007), vegetation composition (1978, 1998 and 2007), and land use and land use change (i.e. Broad Habitat; 1998, 2007). The co-location of the soil carbon measurement with other soil variables and wider square-level explanatory variables provides a unique data source for a full integrated assessment of status and change of soil carbon in GB.

- No large-scale changes in topsoil carbon concentrations, carbon density and stocks at the GB scale between 1978 and 2007 were observed indicating no change in climate regulation through change in this component of our natural environment. Arable systems were the only habitats to show consistent change for both topsoil carbon concentration and carbon density with losses of 10-13% and 5-11%, respectively. This might negatively affect sustainability of food provision services.

- Change in soil pH is the most consistent variable related to a change in topsoil carbon concentration across all three time periods 1978-1998, 1998-2007 and 1978-2007. There is evidence that this change in soil pH is linked to the large-scale declines in sulphur dioxide dry deposition observed in many locations. It is a negative relationship i.e. an increase in soil pH or decrease in soil acidity is related to reduced topsoil carbon concentrations. It is not known whether this is due to reduced plant production above / below-ground or increased soil organic matter decomposition rates. Irrespective of this uncertainty, the result clearly identifies the need to consider the impact of air pollution control policies on soil carbon concentrations and highlights the need to develop integrated monitoring, research and modelling approaches across policy areas.

- Some evidence of associations with change in topsoil carbon concentration were also observed for nitrogen deposition (+ve) and some indicators of a warmer climate (-ve) as originally hypothesised. However, these were either only observed for some time periods or dropped out with the inclusion of other variables. For the 1998-2007 period, soil moisture was the strongest association (+ve) observed however, no association was observed with rainfall. One possibility is increased soil moisture results in swelling of soil and thus sampling of shallower and more carbon-rich layers. No, or only weak, evidence was observed for associations which were linked to many other hypotheses

tested i.e. change in vegetation nutrients, successional and moisture status, plant species richness, soil invertebrate number and taxa.

- Overall, these results demonstrate the potential importance of broad-scale drivers such as air pollution and climate trends on topsoil carbon changes at the broad GB scale. Analyses reported in Chapter 7 additionally indicate that there may be many associations specific to individual habitats or to subsets of the data which cancel out and/or fail to achieve significance at the broad GB scale reported here.

3.1 Introduction

Soil provides many functions including the regulation of nutrient and water supply for food and fibre production, filtering and storage of water thus helping in the provision of water supply, habitat for biodiversity and storage of carbon which helps mitigate the effects of climate change. One soil indicator often linked to the supply of many of these functions is soil carbon or organic matter content. Thus status and change in soil carbon is included in most soil monitoring programmes in Great Britain (Emmett *et al*. 2006). However, as soil carbon pools are large relative to the magnitude of changes which occur on an annual timescale, the measurement of change in carbon content is challenging. There have been a range of reviews to synthesise the evidence currently available but the range of soil types, climate conditions, co-occurring and potentially interacting drivers such as land management, climate change and atmospheric deposition, have limited the conclusions which can be drawn when combined with methodological limitations currently employed (e.g. Schils *et al*. 2008).

Within Great Britain (GB) there is now a series of soil monitoring programmes which have reported on changes in soil carbon or organic matter content. These programmes are focused on specific habitats or countries of GB as a whole. A synthesis of these findings has been collated for the Review of Transboundary Air Pollution (RoTAP) report[17] currently under review. Ongoing work is now focused on understanding observed changes since only by understanding past change can models be validated and advice provided on options for managing soil carbon for different functions.

A range of potential factors responsible for soil carbon change have been proposed, including climate change (Davidson and Janssens, 2006; Heimann and Reichstein, 2008), nutrient deposition (Magnani *et al*. 2007; Pregitzer *et al*. 2008), management practices (Jones and Donnelly, 2004; Jones *et al*. 2006), increasing atmospheric CO_2 concentrations (Jastrow *et al*. 2005), and land use and land use change (Guo and Gifford 2002). However, at the large scale the attribution of change is complicated by the presence of multiple drivers which can simultaneously affect the balance of carbon inputs (largely plant biomass) and outputs (mostly microbial respiration). Climate change is

[17] http://www.rotap.ceh.ac.uk/

thought to be unlikely to be the only cause of previously reported changes in topsoil carbon concentrations, since changes in temperature and rainfall across GB since 1978 have been insufficient to cause large-scale changes in mineral soil carbon concentration or carbon density (Smith *et al*. 2007).

3.2 Biophysical Measurement

Countryside Survey (CS) is an integrated national monitoring programme in which vegetation, topsoil, water and land use measurements are made across GB in the same locations using a stratified random sample of 1km x 1km squares (Carey *et al.* 2008). The soil component of CS is unique as topsoil carbon concentrations (0-15cm) have been measured at three time points (1978, 1998 and 2007) together with topsoil bulk density (2007), a range of other soil parameters (1978, 1998 and 2007), vegetation composition (1978, 1998 and 2007), and land use and land use change (i.e. Broad Habitat; 1998, 2007). These results are now under review by an open process in the journal Biogeosciences (Chamberlain *et al.* in review). The co-location of the soil carbon measurement with other soil variables which may determine the sensitivity of soil carbon to drivers plus potential drivers such as land use and management and change in vegetation composition provides a unique data source for a full integrated assessment of status and change of soil carbon in GB.

The biophysical measurements of relevance for soil carbon (0-15cm) in CS is built upon a series of measurements of loss-on-ignition which is converted to topsoil carbon concentrations, carbon density and stock (Emmett *et al.* 2010). All methods are described in the CS Soils Technical Report (Emmett *et al.* 2008) and the CS Soils Report (Emmett *et al.* 2010).

3.3 Rationale for selection

Countryside Survey is currently the only soil monitoring programme which has reported change in topsoil carbon concentration and carbon density at three time points for the whole of GB. This time series combined with co-located soil, vegetation and land use data sources provides a unique opportunity to carry out an integrated assessment without the usual problems of combining data collected at different scales and locations. In addition, there is interest by a wide range of government departments and agencies in identifying the potential role of soil carbon sequestration in contributing to government targets to reduce greenhouse gas emissions. For example, the Soil Strategy for England states “UK soils contain 10 billion tonnes of carbon – more than in all the trees in the forests of Europe (excluding Russia). This is equivalent to more than 50 times the UK’s current annual greenhouse gas emissions. Well-managed soils have the potential to sequester more carbon in future, but more needs to be done to understand and optimise this process. Soils can also help us to adapt to a changing climate and, through changing management practices, increase our ability to deal with changes in our climate.”

Specific options to enhance soil carbon storage have been explored in several recent reviews[18]. This level of current interest in soil carbon for mitigating climate change contributed to the decision to focus on soil carbon as a potential indicator from CS of an important ecosystem service.

3.4 How is soil carbon linked to ecosystem services?

The link between soil carbon storage, ecosystem function, ecosystem service and benefit is illustrated following the Haines-Young and Potschin (2007) cascade (Fig. 3.1). In contrast to other services where supply of a service may not equate to uptake/use and thus illustrates potential supply only, the benefit of carbon storage in soils is realised globally as carbon dioxide is well mixed in the atmosphere and climate regulation delivered without any human-assisted exploitation of soil carbon concentration. Soils are one source of this service with plant biomass and the ocean also providing significant regulation of carbon dioxide and other greenhouse gas emission concentrations in the atmosphere.

18 http://wales.gov.uk/topics/environmentcountryside/farmingandcountryside/farming/landuseclimatechangegroup/?lang=en
http://www.fcrn.org.uk/fcrnEvents/seminars/soilcarbon/pdfs/FCRN_SoilCarbon_summary.pdf

Linkage of soil carbon to other soil functions including the provision of food has been reviewed in a series of reports by the Environment Agency (2002, 2006a,b and 2008a,b). Soil organic matter content is an indicator of Defra’s Sustainable Food and Farming Strategy reflecting the perceived importance of soil organic matter for food provision.

There are several limitations in the biophysical measurement reported in Countryside Survey which impact on estimates of change in the ecosystem service provided. These are:

(a) Sampling 0-15cm:

Sampling was largely limited to the top 15cm and therefore changes in lower horizons cannot be quantified. This may result in under or over-estimation of soil carbon change. Ideally monitoring would extend to 1m depth.

(b) No measure of erosion losses

Surface erosion losses will effectively result in sampling of lower soil horizons in later surveys. This could result in an under-estimation of soil carbon loss.

(c) Variable depth sampling in peat soils

By definition, the organic horizon of peat soils is at least 40 cm deep, and therefore a loss or gain of organic matter from the surface is unlikely to be reflected in a 0-15 cm fixed-depth sample. Additionally, peats expand and contract depending on their moisture content, and a fixed-depth sample will therefore have a greater or lesser carbon content depending on the moisture content at the time of sampling. Two possible methods can overcome these problems: sampling to a dateable horizon or sampling to a fixed point. Current methodology is likely to have limited impact on estimation of change in carbon concentration but may under or over-estimate change in carbon density and stocks depending on the moisture status at time of sampling.

However, there are many benefits of the CS dataset relative to others available including:

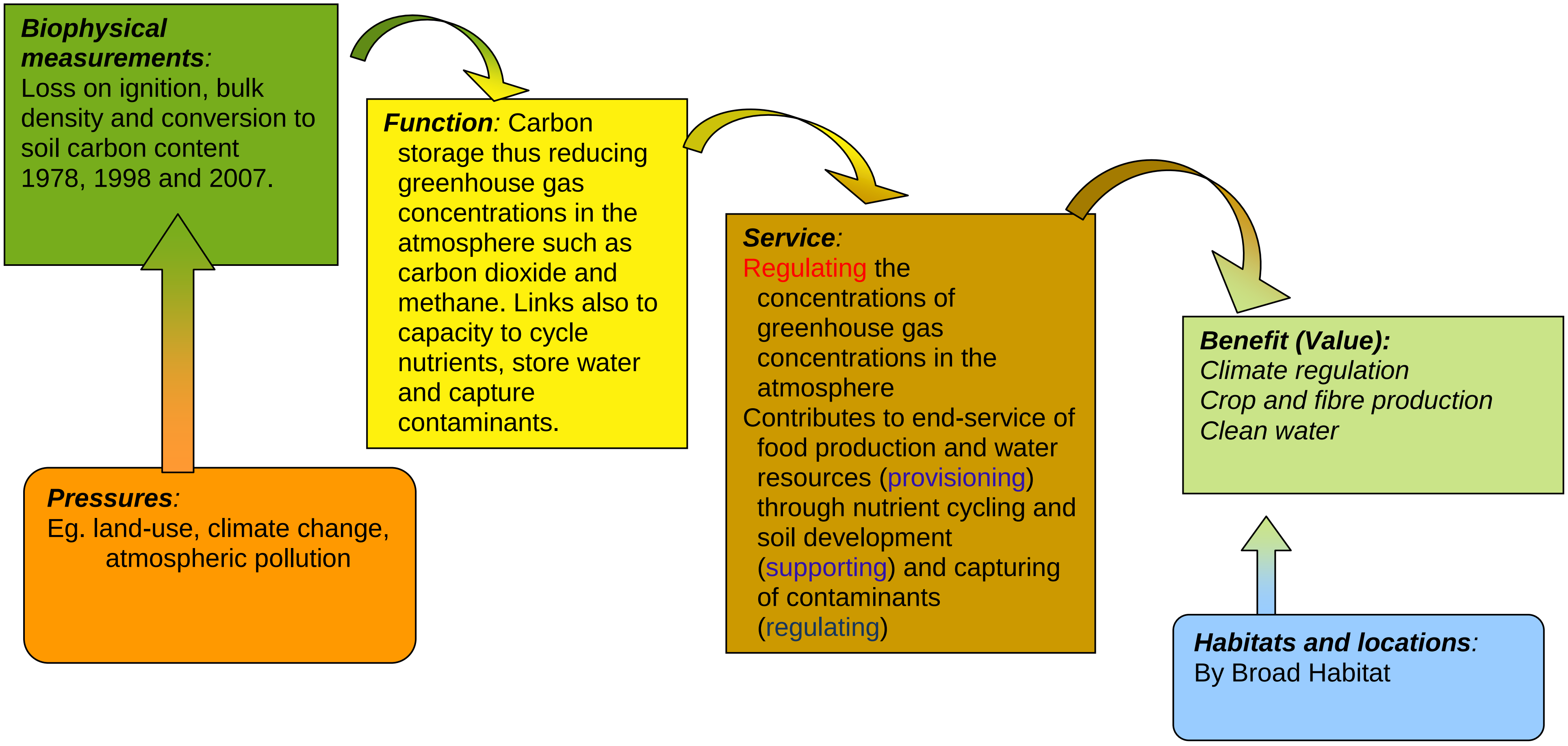
(d) spatial sampling covering Scotland, England and Wales and all Broad Habitats

(e) highly efficient sampling design (see Black *et al*. 2009)

(f) consistent analytical methodology for all surveys

(g) inclusion of bulk density measurements enabling change in soil carbon density as well as content to be calculated rather than estimated.

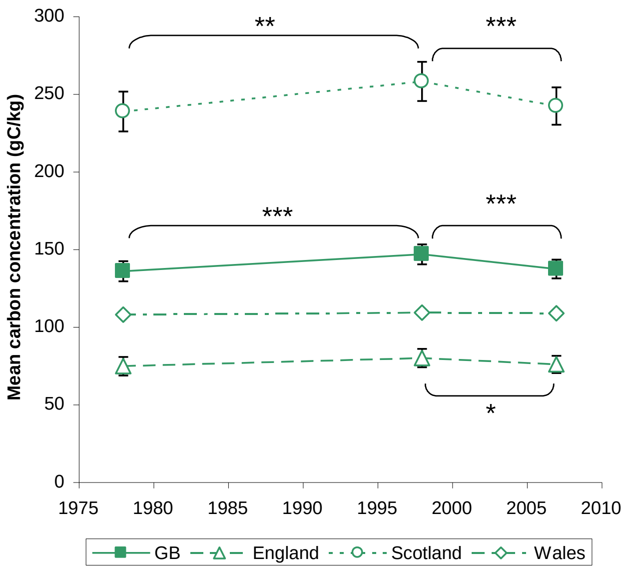
Figure 3.1: The ecosystem service cascade for soil carbon (after Haines-Young and Potschin 2007).



3.5 Current status and trends across GB

Change in topsoil carbon concentration and content (0-15cm) is reported in full in the CS Soils report (Emmett *et al*. 2010). In brief, a significant small increase in topsoil carbon concentration (0-15cm) was observed between 1978 and 1998 and a small decrease between 1998 and 2007. No overall significant change was observed between 1978 and 2007 (Fig. 3.2).

Figure 3.2: Change in soil carbon concentration (0-15cm) for GB and individual countries over time. Standard errors are indicated. Significant differences (<0.001, **p<0.01, *p<0.05) are shown between years bracketed.***

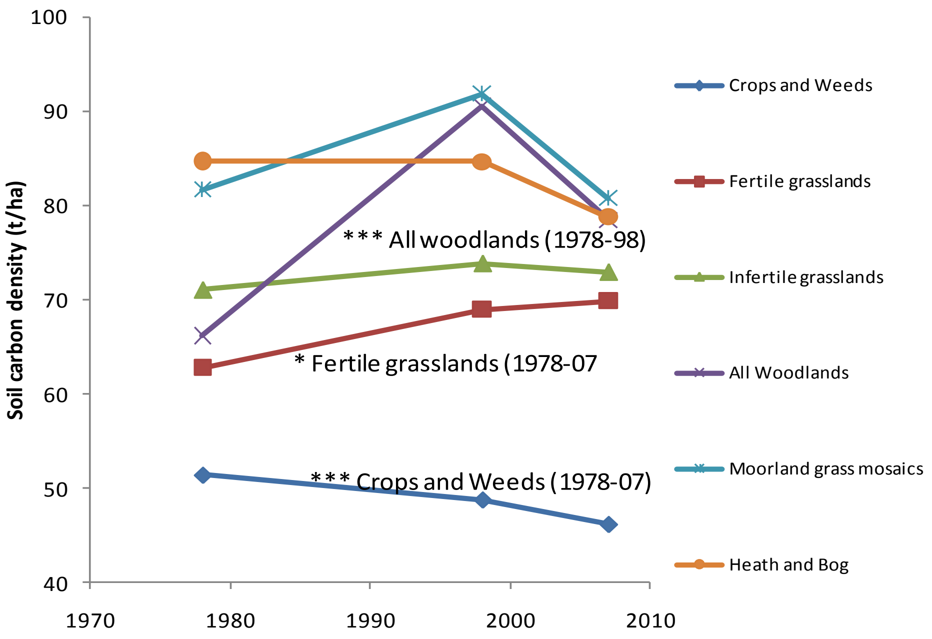


An overall mean change of zero can result from the averaging out of a spatially distributed pattern of increases and decreases that could in turn be explainable in terms of the varying severity of different drivers. In addition, if soil carbon increased or decreased to the same extent everywhere attribution to driving variables would be problematic because there would be no way of contrasting the relative effect of zero or low driver levels with larger impacts linked to more intense operation of drivers (Stow et al 1998).

Significant changes were observed both between the three survey dates and within some habitat types. When corrected for bulk density, the change in topsoil carbon density (0-15cm) where Aggregate Vegetation Class (AVC) has not changed was only significant for All Woodlands 1978 –1998, Fertile Grassland 1978 – 1998 and Crops and Weeds 1978 – 2007 (Fig. 3.3). AVC is defined as a high level grouping of vegetation types produced from a quantitative hierarchical classification of the different plant species found in the original Countryside Survey sample plots (Bunce et al 1999a).

To remove the effects of large-scale vegetation/land use change, we also estimated topsoil carbon concentrations in plots which were sampled in all three Surveys, and in which the AVC did not change over time. Since the AVC of the plots is only known for the Survey years we cannot rule out changes in the intervening years e.g. rotation between arable and grassland systems. Therefore plots where the AVC has not changed are plots in which the vegetation composition has been largely consistent in 1978, 1998 and 2007. Only 405 plots had consistent AVCs in the three Surveys for GB. Because of this there were insufficient samples in the Tall Grass and Herb AVC, and soils in this category were ignored. Additionally, the Lowland and Upland Woodland AVCs were grouped into one category -All Woodlands - due to the small number of samples in each category individually. Nevertheless, trends in these plots were broadly consistent with those observed for the whole dataset suggesting that shifts in AVC (i.e. land use change) are unlikely to be a major factor determining soil carbon concentration changes at a national scale over time (Chamberlain *et al*. In review).

Figure 3.3: Change in topsoil carbon density over time for Aggregate Vegetation Classes which have remained constant between surveys. Significance levels: * P<0.05, ** P<0.01, *** P<0.001 are indicated.

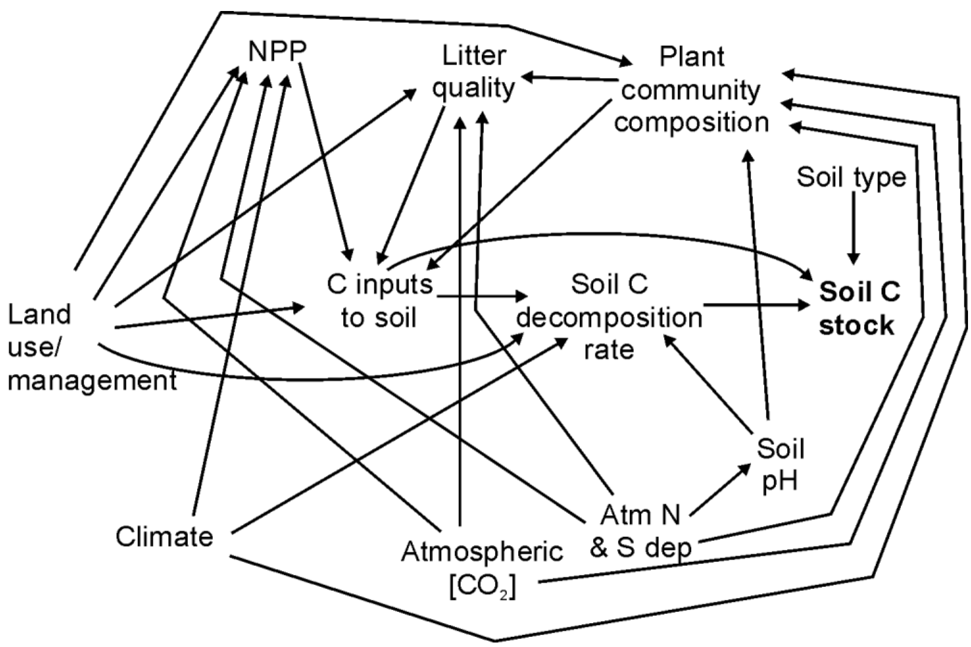


3.6 Attribution of change

Hypotheses of drivers of change in topsoil carbon concentration

Possible factors that may have contributed to changes in topsoil carbon concentration (0-15cm) since 1978 include: changes in land management and/or cover; changes in atmospheric pollution; climate changes; and responses to changing weather patterns. Different factors may be operative in different habitat types. The complexity of potential factors are illustrated in Figure 3.4.

Figure 3.4: The possible factors which may affect topsoil carbon stock and their interactions.



Specific hypotheses which can potentially be investigated are:

Land cover and management:

- Change e.g. from grassland to arable, or grassland to woodland or change in intensity of management causing change in a range of physical, chemical and biological changes which would affect the balance between carbon inputs and losses

Climate:

- Temperature e.g. higher temperature increasing plant carbon fixation growth and decomposition
- Hydrological change e.g. wetter winters increasing plant production in drier soils or and slowing decomposition in peat soils
- Higher atmospheric CO_2 concentrations e.g. promoting increased plant production
- Longer growing season e.g. promoting increased plant production
- Extreme weather events e.g. 1987 storm either introducing unusually large amounts of new litter into the soil, or conversely, new gaps promoting new herbaceous biomass, more easily decomposed and therefore stimulating carbon loss from enhanced microbial activity and more rapid mineralisation.

Air pollution:

- Nitrogen deposition e.g. increased plant growth in nitrogen-limited habitats and variable effects on decomposition rates
- Reduced sulphur deposition e.g. accelerated decomposition due to reduced soil acidity

Plant species composition:

- Change in plant species composition e.g. changes in plant production, litter quality, water use efficiency

In addition to these direct factors which may result in changed topsoil carbon status there are inherent sites and topsoil properties which may result in greater sensitivity to these factors e.g.:

- Vegetation type e.g. change in nitrogen deposition may be more critical for changing carbon inputs in vegetation types characterised by nitrogen limitation
- Soil type e.g. organic soils may be more sensitive to a change in hydrological conditions
- Critical load class e.g. inherent buffering capacity of soils to change in acidic deposition change and thus impacts of acidic deposition

Some of the above hypotheses cannot be tested due simply to lack of data. Others can only be tested indirectly using proxy variables or indirect measures. The variables used in statistical analysis are described in detail in Appendix 3.1. A summary is presented in Table 3.1.

Table 3.1: Summary of variables used in statistical analysis of drivers of change in CS topsoil carbon content (0-15cm) measurements 1978, 1998 and 2007.

Categories	Variables included in analysis
Soil variables	
Physical	Soil group (1998); Hand texture (2007); Soil moisture (1998, 2007)
Chemistry	Soil pH average and change (1978, 1998, 2007), topsoil carbon concentration average and change (1978, 1998, 2007)
Biological	Invertebrates (1998, 2007); Average and change in: Mites, *Collembola*, Shannon diversity, No. of taxa Soil bacteria (2007); Three PCA diversity indices, Shannon diversity
Acidification sensitivity	Critical Load Class
Atmospheric deposition	
Nitrogen deposition (NHx, NOy)	Average (2004-2006). Change assumed to have been small.
Sulphur deposition (SOy)	Wet, dry and total + change for each (1970, 1995, 2005)
Habitat specific	Woodland and arable of above
Climate variables	
Temperature	Average annual temperature
Rainfall	Average annual rainfall
Climate trends	Trend in average annual temperature; Trend in annual rainfall; Long-term trend in growing degree days; Long-term trend in growing season length
Land management, vegetation indices and species	
Major axes of change	Three indices related to management intensity, fertility and moisture (improvement on Ellenberg indices)
Species richness	Total higher plant species richness
Cover change	Indicators of change: to/from woodland, arable to/from grassland
Disturbance	1987 storm track

Statistical methods

There are a range of issues which need to be considered concerning the statistical analysis of CS soils data. These are more fully discussed in Appendix 3.2. In summary, the CS dataset is hierarchical since plots are nested within squares. In common with approaches taken in other chapters a mixed modelling approach has been employed (see Chapters 2, 4, 5 and 8). For exploratory purposes some statistics, e.g. correlations and means of plot values, have been derived ignoring the nested structure of the data hierarchy but in all cases significance tests have been derived taking account of the hierarchical structure.

Preliminary / Exploratory analyses

Initially each factor was investigated individually to understand the relationship of factors with topsoil carbon change (0-15cm) in isolation. These are a combination of factors which co-vary spatially with change in topsoil carbon and factors which vary temporally with topsoil carbon (Table 3.2).

Table 3.2: Correlations between change in topsoil carbon concentrations and soil properties, air pollution and climate variable: (i) spatial relationships (ii) temporal relationships (* P< 0.05, ** P<0.01, *** P<0.001, Blank= not significant).

	1978 – 1998	1998 – 2007	1978 – 2007
Spatial relationships			
Average pH	-0.10*		-0.11*
Average SOy (total)			-0.14*
Average annual temperature			-0.14*
Shannon diversity soil bacteria	-0.16***		
Average Invertebrate (Total catch; Mites; Collembola; Total taxa)	0.11 - 0.12*		0.09 – 0.12*
Temporal relationships			
Change in pH	-0.09**	-0.08*	-0.09**
Change in moisture		0.24***	
Change in SOy (total)			0.14*
Change in SOy (dry)			0.14*
Long-term trend in growing degree days	-0.19**		
Change in vegetation nutrient status (DCA axis 1) (NB: An increase in score is associated with a low nutrient status)			0.08*

Spatial relationships:

a) A negative relationship between topsoil carbon concentration change and average pH is observed for 1978-98 and 1978–2007. This may be linked to vegetation types and land management associated with the less acid more fertile soils resulting in a greater risk of topsoil carbon loss. For example, Crops and Weeds vegetation type was the habitat which displayed consistent significant topsoil carbon concentration loss over time.
b) A negative relationship is observed between change in topsoil carbon concentration and average annual temperature. Decomposition is thought to be more temperature sensitive in northern latitudes which may be one possible explanation of this finding i.e. a trend towards warmer conditions would have greater impact in colder climates.
c) There was a positive relationship between numbers of soil invertebrates in 2007 and change in topsoil carbon concentration 1978-1998 and 1978-2007. However, no general association was observed between change in soil invertebrate numbers or diversity between surveys and topsoil carbon concentration when both measures were available (1998-2007). Habitat specific analyses do, however, show a relationship for Dwarf Shrub Heath and Acid Grassland (Chapter 7).

Temporal relationships

a) A consistent negative relationship with change in soil pH is observed (Fig. 3.5) and a positive significant relationship between change in topsoil carbon concentration and change in sulphur dioxide dry deposition (Fig. 3.5) and total deposition (Table 3.2). As sulphur deposition has declined across GB since its peak in the early 1970s, this is consistent with the relationship observed for change in soil pH i.e. the largest reductions in sulphur deposition (which would result in a greatest increase in soil pH) are associated with smaller increases or reduced topsoil carbon concentrations. There was no association with reduced or oxidised nitrogen deposition.
b) Change in moisture is the soil variable most strongly associated with change in soil carbon concentration 1998 – 2007. The positive relationship may indicate relief from moisture limitation on decomposition processes accelerating carbon loss from the soil.
c) A negative relationship is observed between change in topsoil carbon concentration and the long-term trend in growing degree days (Fig. 3.5). This is consistent with a greater increase in decomposition rates of soil organic matter relative to plant production rates due to warmer temperatures. No relationship to the 1987 storm damage track was observed.
d) A positive change in vegetation nutrient status (DCA1) between 1978 and 2007 was positively associated with change in topsoil carbon concentration 1978 – 2007 (Fig. 3.5). No association with total plant species richness was observed. This perhaps suggests that whilst no relationship was detected between nitrogen deposition and topsoil carbon concentrations, effects on topsoil carbon may be expressed if

vegetation composition changes. An increase in topsoil carbon appears to be associated with nutrient poor vegetation thus a shift towards more nitrogen-loving plants is associated with a reduction in topsoil carbon concentrations. Interestingly, within the Arable Broad Habitat (which is the only Broad Habitat with a significant change in topsoil carbon concentration (-ve) there is a significant negative relationship with both change in vegetation nutrient status (DCA1) and vegetation shading/successional status (DCA2) suggesting these relationships may be habitat specific.

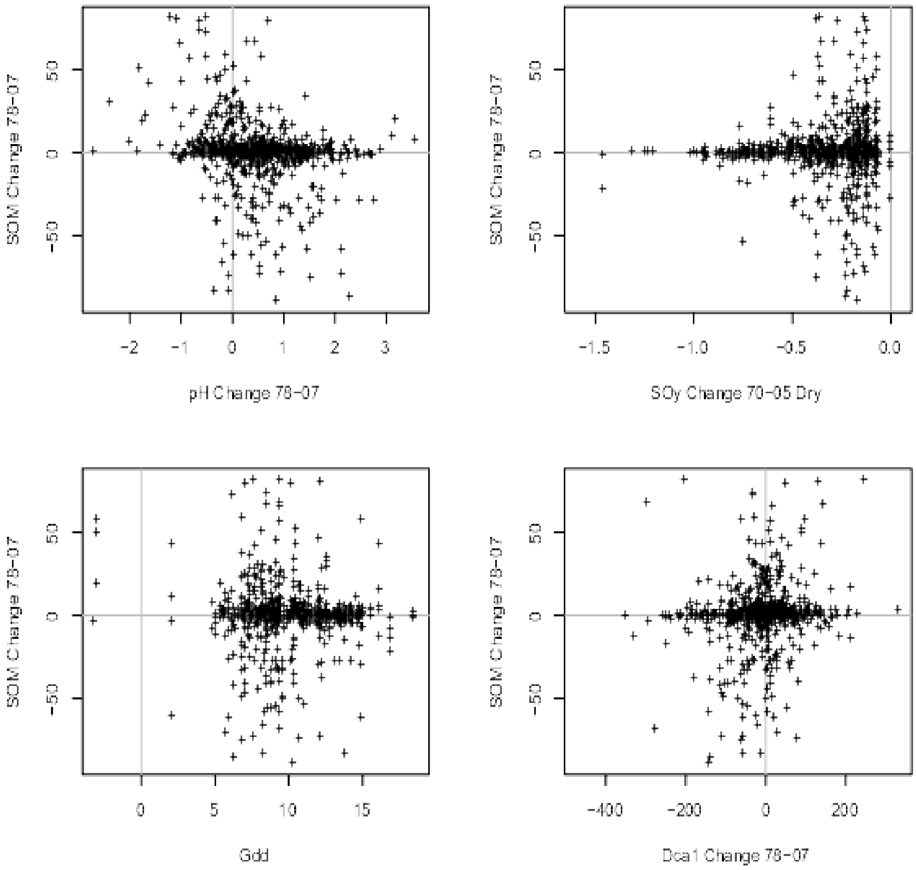


Figure 3.5: Significant relationships between change in topsoil organic matter (SOM) between 1978 and 2007 and changes in soil pH and sulphur dioxide dry deposition, the long term trend in growing degree days and the positive change in vegetation nutrient status (DCA1) over the same period.

Multiple variable models

The following section includes analysis of the relationship of variables concurrently. A mixture of techniques was used to derive the final reported models but all based on the same mixed model structure with plots nested into 1km squares. Starting with all variables in the model backwards elimination was used to delete non-significant variables and to give a minimal model. A second set of analyses were performed starting from the same set of variables but with those variables measured at only one timepoint removed. This provides a comparison by which to judge possible regression to the mean effects. For the 1998-2007 period a third set of analyses was obtained with change in moisture removed as well as the single timepoint variables. Soil moisture is the variable most closely associated with topsoil carbon concentration and it has been suggested that its effect on change arises from sampling effects due to soil expansion. Furthermore it is only available for the 1998-2007 period and it is therefore of interest to compare results without this variable to the results from other periods. For each final model additional checks on omitted variables were then made to see if they contributed to the interpretation or improved the fit of the model before a final choice was made. Particular emphasis was placed on testing plausible variables and interactions.

1978-1998

Three variables are retained for the period 1978-1998: NOy (+ve), trend in growing degree days (-ve) and soil group (Appendix 3.1). The first two variables are only just significant ($p=0.04$ and $p=0.049$) and the third is defined in 1978 so may be included because of regression to the mean. If soil group is excluded the final model contains NOy (+ve) and trend in growing degree days (GDD) (-ve) which are both slightly more significant suggesting that they were not included in the first model as an indirect effect of the inclusion of soil group. The fitted estimates from these models imply that the probability of an increase in topsoil carbon concentration in high NOy deposition areas and a reduction where there is an upward trend in growing degree days (i.e. a lengthening of the growing season).

1998-2007

For this period the final model starting from the full set of selected variables only has two variables, change in soil moisture (+ve) and microbial diversity (-ve) (Appendix 3.1). The inclusion of microbial diversity is suspicious given that it was only measured in 2007 and no measure of change is available. If accepted at face value it suggests that increase in topsoil carbon concentration is greatest (or decrease least) where microbial diversity is low. The final model when variables not measured at both timepoints are excluded has three variables, change in topsoil moisture (+ve), GDD (-ve) and trend in growing season length (GSL) (+ve). If topsoil moisture is removed from this model the remaining two variables become not significant. When starting from a variable set which does not include moisture change, the final model just includes change in pH (-ve).

1978-2007

The final model that is reached for the overall period of 1978-2007 is the same whether the single timepoint variables are excluded or included. Two variables are selected: change in pH (-ve) and Aggregate Vegetation Class (AVC). The estimated values for AVC categories show a drop in arable plots and a rise in all other categories with the greatest rise being for Moorland/heathland plots. Change in pH is not significant if AVC is removed but AVC remains significant if change in pH is removed. Growing degree days, which was significant for some of the analyses in both of the component periods, is significant if included in a model with just pH change as the other explanatory variable but not if it is the only variable or if AVC is included.

Taken together the results of the preliminary analyses and the multivariate model fitting are suggestive but not conclusive. Clearly change in topsoil pH emerges as an important variable both in the individual analyses and the mixed model approach for the whole 1978–2007 period. It seems likely from the preliminary analyses this is linked to a change in SOy deposition although this did not emerge in the final mixed model possibly due to lags in topsoil pH response or the response of different topsoil communities to changing acidity which is captured in the AVC response. This is the first time a national scale change in topsoil carbon concentrations has been linked to acidic measures to our knowledge.

Of the deposition measures only NOy in the 1978-1998 period was significant. Since this variable measures levels and not change it is likely that its inclusion in the model may simply be a reflection of the geographical association of deposition and topsoil carbon concentration but is an intriguing finding which should be explored further.

For the most recent CS period of 1998-2007 it is clear that change in topsoil moisture is the variable most strongly associated with topsoil carbon concentration change over time. It is unfortunate that topsoil moisture was not recorded for the 1978 survey. The question remains however, as to the extent to which this is a sampling effect arising from topsoil expansion and contraction under different moisture conditions and whether it could account for the rise in topsoil carbon concentration values from 1978 to 1998 as well as the subsequent fall. Since topsoil moisture changes are, at least in part, influenced by rainfall it might be hoped that rainfall measures could be used to explain changes in topsoil carbon concentration when topsoil moisture was not available, but this does not seem to be the case; rainfall did not come out as significant in any of the analyses.

Growing degree days is significant in both individual periods and is significant in the overall period (if included in the model with pH change) suggesting a possible role for climate change in topsoil carbon concentration change.

Overall, variables measuring change in vegetation were not found to be significantly associated with change in topsoil carbon concentration, although the habitat specific analyses in Chapter 7 do suggest there may be some relationship within Arable and Neutral Grassland and for butterfly food species in Bog Habitats. Similarly although topsoil biota were strongly associated with topsoil carbon concentration there was no indication that changes in soil biota were associated with change in topsoil carbon concentration at the GB level, although clearly the ability to test this is limited since microbial data were only available for 2007 and invertebrate data only for 1998 and 2007. Again habitat specific analyses in Chapter 7 suggest some relationship may be present in two Broad Habitats indicating that global broad scale results may conceal an underlying fine structure. More detailed work needs to be done to understand the complicated interactions between variables such as vegetation, rainfall, topsoil moisture, pH and GDD and the resulting effects in terms of changes in topsoil carbon concentration.

Revisiting the hypotheses reveals the following:
- Change in land cover and management (no evidence)
- Change in climate (some evidence)
- Change in air pollution (some evidence)
- Change in plant species composition (some evidence)

3.7 Conclusions

No large-scale changes in topsoil carbon concentrations, carbon density and stocks at the GB scale between 1978 and 2007 are observed indicating no change in climate regulation through change in this component of our natural environment. Arable systems are the only habitats to show consistent change for both topsoil carbon concentration and carbon density with losses of 10-13% and 5-11%, respectively. This may negatively affect sustainability of food provision services.

The relationship between spatial and temporal variability in topsoil carbon concentration change between surveys 1978, 1998 and 2007 across GB and a range of potential direct and indirect explanatory variables indicate:

(a) Change in soil pH is the most consistent variable associated with change in topsoil carbon concentration across all three time periods 1978-1998, 1998-2007 and 1978-2007. There is evidence that this change in soil pH is linked to the large-scale declines in sulphur dioxide dry deposition observed in many locations. It is a negative relationship i.e. an increase in soil pH or decrease in soil acidity is associated with reduced topsoil carbon concentrations. It is not known whether this is due to reduced plant production above / below-ground or increased soil organic matter decomposition rates. Irrespective of this uncertainty, the result clearly identifies a need to consider the impact of air pollution control policies on topsoil carbon concentrations and highlights the need to develop integrated monitoring, research and modelling approaches across policy areas.

(b) Some evidence of relationships with change in topsoil carbon concentration were also observed for nitrogen deposition (+ve) and some measures of a warmer climate (-ve) as originally hypothesised. However, these were either only observed for some time periods or dropped out with the inclusion of other variables. For the 1998-2007 period, soil moisture was the strongest association (+ve) observed however no association was observed with rainfall. One possibility is that increased soil moisture results in swelling of soil and thus sampling of shallower and more carbon-rich layers. No or only weak evidence was observed for associations which were linked to other hypotheses i.e. change in vegetation nutrient, successional and moisture status, plant species richness, soil invertebrate number and taxa.

Overall, these results demonstrate the potential importance of broad-scale drivers such as air pollution and climate trends on topsoil carbon changes at the broad GB scale. Analyses reported in Chapter 7 indicate that there may be associations which are specific to individual habitats which cancel out at the broad GB scale reported here.

Chapter 3: Online appendices

Appendix contains:

3.1: Descriptions of explanatory variables used in the attribution analysis.
3.2: Notes on analytical and statistical issues.
3.3: Statistical test results.

Chapter 4: Appropriate diversity; a ‘cultural’ ecosystem service

S.M. Smart, LC. Maskell, P. Henrys,

Summary

- *‘Appropriate diversity’* is a new term coined for this study to describe the abundance of desirable or undesirable plant species in British habitats. These species help measure how nature conservation value varies from place to place. In the UK the primary objective of nature conservation has been stated as “to ensure that the national heritage of wild flora and fauna and geological and physiographic features remains as large and as diverse as possible, so that society may use and appreciate its value to the fullest extent” Nature Conservancy Council (1989). Hence, appropriate diversity is an indicator of a cultural ecosystem service.

- Appropriate diversity was measured by the species richness and cover of Common Standards Monitoring (CSM) Indicator species in Countryside Survey (CS) vegetation plots in 1998 and 2007. Species selected were those referable to Biodversity Action Plan Priority Habitats[19] within common Broad Habitats in Britain.

- Analyses of current status and recent change between 1998 and 2007 showed that in most Broad Habitats there were decreases in mean richness of both positive and negative CSM Indicator species richness consistent with a general reduction in species diversity in Countryside Survey plots between 1998 and 2007. Negative CSM Indicator cover increased the most in the linear Broad Habitat types.

- Correlations between change in negative indicator cover and potential drivers were most consistent with expectation. Long-term climate warming since 1980 was associated with increased negative CSM Indicator cover in Neutral Grassland, Boundary and Linear Features and Rivers and Streams while reduced nitrogen deposition also favoured increased negative CSM Indicator cover in a number of Broad Habitats. These relationships provide important evidence of the impact of anthropogenic factors outside of the control of site managers on CSM Indicator species abundance in the wider countryside.

- Correlations between negative and positive CSM Indicator richness and potential drivers were often inconsistent with expectation and may reflect the diversity of traits within each pool of indicators such that responses among species were of mixed direction.

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[19] See http://www.ukbap-reporting.org.uk/plans/national.asp for current action plans.

- An ongoing issue is the lack of finely-resolved data on the history of management impacts linked to management under agri-environment schemes. This remains a major obstacle to quantifying large-scale progress in maintaining and restoring British habitats and their ecosystem services. An important implication of this inability to track positive management status is that attribution analyses may over-represent negative impacts purely because of absent or poor quality data.

- An example was produced of an established but rapidly developing Europe-wide approach to modelling the impact of multiple drivers on biodiversity. Two niche models were developed for ombrotrophic *Sphagnum* species in Britain and hence for an ecosystem dominant and positive CSM Indicator of appropriate diversity across an extensive UK Priority Habitat.

- Scenarios of predicted climate change and atmospheric pollutant deposition impacts (sulphur and nitrogen) were applied to upland bogs across the UK to simulate expected change in habitat suitability between 2020 and 2050.

- Despite high uncertainties on model parameters a consensus between the models could be inferred enabling initial assessment of areas most at risk of climate change and pollution. Scenario induced changes were however, extremely small relative to other sources of variation in the predictions.

- The example considered a positive CSM Indicator species group for one Priority Habitat in Britain. A much larger range of presence-based niche models now exist enabling most CSM species to be modelled in a similar fashion.

4.1 Introduction

The obvious value placed by society on wild species, habitats and biodiversity means it ought to be easy to categorise the ecosystem service provided by our natural heritage of wild flora and fauna. Tackling the problem shows it is not straightforward. Whilst biodiversity in general can be viewed as underpinning all ecosystem services, the totality of biodiversity appears too broad to make this a useful treatment since it includes undesirable species as well as subsets of species that are more obviously linked to other specific services and are better treated as such e.g. nectar plants or crop species and their wild relatives. In addition some groups of species, such as bacteria and mychorrhizal fungi which have not generally been the subject of conservation action and concern contribute to delivery of a number of ecosystem services (e.g. water and soil purification services).This chapter deals with species desirable from the point of view of their nature conservation value. The species valued under this banner might also contribute to pollination or soil carbon storage, but here the focus is on those species that are conserved on nature reserves and reflect the value we place on the parts of our landscape that are viewed as more wild and natural, including the species that depend upon these habitats for their survival. This chapter analyses current status, trends and possible causes of recent change in groups of these species defined by the Priority Habitat types for which they are considered characteristic. The measures used in this chapter are measures of ‘appropriate diversity’ since different species contribute to the conservation value of different habitats. We also conclude that appropriate divesity is best considered a cultural service in the sense of the Millenium Ecosystem Assessment.

Even if it is difficult to assign the conservation of nature to a service category, the policy recognition of the value of wild flora and fauna is very well developed. For example assessment of the condition of our SSSI relates directly to national Public Service Agreement targets[20], whilst targets for the conservation of our wild flora and fauna are clearly set out in the UK Biodiversity Action Plan (BAP).

The chapter ends with an example of how modern statistical techniques can be used to model the niche of species of high conservation value. These models were then used to explore the possible impact of future environmental change on prospects for species’ survival in the coming decades.

4.2 Biophysical measurement

‘Appropriate diversity’ is a term coined for the purposes of this study. It is a measure of the number of species in Countryside Survey (CS) plots that are either desirable or undesirable in relation to explicit criteria for assessing the condition of GB Priority Habitats. These criteria and the identity of the Indicator species were taken, from Common Standards Monitoring (CSM) Guidance[21] or supplemented by other indicator species lists for Broadleaved, Mixed and Yew Woodland and Arable and Horticulture since these lack CSM lists. The identity of the species counted in each plot reflected the identity of the BAP Priority Habitats that fall within the scope of the Broad Habitat in which the plot was located. The definition of many Broad Habitats means that they consist entirely of their constituent Priority Habitats. For example, Upland and Lowland Heath together define the Dwarf Shrub Heath Broad Habitat. However, some Broad Habitats may include areas that would not be considered as Priority Habitat on the grounds of their species composition. This could include areas of Inland Rock or Broadleaved Woodland, among

[20] www.defra.gov.uk/rural/protected/sssi/psa.htm
[21] www.jncc.gov.uk/guidance

other examples. In this analysis, the number of CSM Indicator species were counted in each plot within the Broad Habitat irrespective of whether the area in which a plot was located would actually qualify as an associated Priority Habitats. This is consistent with an objective approach to quantifying the diversity of CSM Indicators in the wider countryside both inside and outside designated sites and across habitat patches that are likely to vary in quality and hence appropriate diversity. In addition, in linear plots that represent the Boundaries and Linear Features and Rivers and Streams Broad Habitats, all CSM Indicators from all Priority Habitat lists were counted. This is because these habitats lack CSM Indicator lists but the approach is also consistent with viewing these habitats as potential refuge features where a range of 'desirable' species may persist in the wider landscape yet not within areas that would be defined as part of their associated Priority Habitat (Smart *et al.* 2002, 2006).

Measuring 'appropriate diversity' in Countryside Survey plots

The response variables analysed were either counts of CSM Indicator species (both positive or negative) in each plot or total cover of negative CSM Indicators. Where possible, CSM Indicators were counted in each plot but a number of modifications were applied to account for differences in the distribution of Priority Habitats within Broad Habitats and the occasional absence of CSM lists for some Priority Habitats[22].

Improved Grassland, Bracken and Coniferous Woodland were excluded from the analysis since the first two are not associated with any Priority Habitats and Caledonian Pinewoods were not adequately sampled in the survey. In addition, coastal habitats were excluded since CS does not provide representative coverage. Two Broad Habitats lacked published CSM lists and so alternative lists of 'desirable' species were used. In Arable and Horticulture the count of annual dicotyledonous herbs was applied as a positive indicator reflecting its use as a positive indicator in the relevant Habitat Action Plan[23]. For Broadleaved, Mixed and Yew Woodland, Ancient Woodland Indicators (AWI) (see Kirby *et al.* 2005) were counted in each plot.

In CS, different types and sizes of plots are used to sample different habitats; the most important distinction being between linear habitats and larger areas in fields, enclosures or in unenclosed land. Therefore, different types of plots were involved in the analysis. The Cereal field margin Priority Habitat was represented by A Plots (1 x 100m) recorded along the edge of cultivated fields in 1998 and 2007. In the two linear Broad Habitats; Boundaries and Linear Features and Rivers and Streams, indicators were counted in Linear Plots (1 x 10m). All other Broad Habitats were represented by 2x2m U, Y and X Plots (central 2x2m nest)[24]. Bryophytes were excluded throughout apart from in two instances, *Racomitrium lanuginosum* and *Sphagnum* spp., where recorded data are considered reliable enough for use. *Sphagnum* spp were grouped

[22] See Appendix 4.1 for tabulation of CSM Indicator specie sfrequency in CS plots.
[23] http://randd.defra.gov.uk/Document.aspx?Document=BD1631_6387_FRP.doc
[24] See Carey *et al* (2008) for further details on plot types and vegetation sampling methods.

into crude but useable categories of 'red/thin', 'red/fat', 'green/thin' and 'green/fat'[25].

Examples of indicators of appropriate diversity for each Broad Habitat are given in Box 4.1.

Indicators of 'appropriate diversity' — *Box 4.1*

These are examples from much longer lists available from the sources cited in the main text.

BROADLEAVED, MIXED & YEW WOODLAND: (Ancient Woodland Indicators)
Carex pallescens, C.remota, Galium odoratum, Festuca gigantea, Allium ursinum, Paris quadrifolia, Convallaria majalis, Hyacinthoides non-scripta, Sanicula europaea

ARABLE & HORTICULTURE: (annual dicots)
Aethusa cynapium, Anthemis cotula, Veronica persica, V. hederifolia, Stellaria media, Polygonum aviculare agg., Stachys arvensis, Chrysanthemum segetum, Viola arvensis,

NEUTRAL GRASSLAND: (CSM Indicators)
Positive: *Centaurea nigra, Conopodium majus, Filipendula ulmaria, Persicaria bistorta, Rhinanthus minor, Lathyrus pratensis, Succisa pratensis, Primula veris, Serratula tinctoria*
Negative: *Plantago major, Cirsium arvense, C.vulgare, Anthriscus sylvestris*

ACID GRASSLAND: (CSM Indicators)
Positive: *Calluna vulgaris, Anenome nemorosa, Aphanes arvensis, Anthoxanthum odoratum, Thymus polytrichus, Astragalus danicus, Campanula rotundifolia*
Negative: *Holcus lanatus, Cirsium arvense,Lolium perenne,Trifolium repens, Senecio jacobaea*

CALCAREOUS GRASSLAND: (CSM Indicators)
Positive: *Antennaria dioica, Asperula cynanchica, Briza media, Campanula glomerata, C.rotundifolia, Cirsium acaule, Carlina vulgaris, Geranium sanguineum, Thymus polytrichus*
Negative: *Senecio jacobaea, Cirsium vulgare, C.arvense, Bromopsis erecta, Plantago major*

FEN, MARSH & SWAMP: (CSM Indicators)
Positive: *Anagallis tenella, Angelica sylvestris, Berula erecta, Caltha palustris, Carex rostrata, C.dioica, C.flacca, C.hostiana, Juncus acutiflorus, Lychnis flos-cuculi, Mentha aquatica*
Negative: *Epilobium hirsutum, Holcus lanatus, Ranunculus repens*

DWARF SHRUB HEATH: (CSM Indicators)
Positive: *Andromeda polifolia, Empetrum nigrum, Calluna vulgaris, Drosera spp, Erica spp, Eriophorum angustifolium, Myrica gale, Viola riviniana, Vaccinium spp, Galium saxatile*
Negative: *Agrostis stolonifera, Anthoxanthum odoratum, Cirsium arvense, Holcus lanatus, Junus effusus, Pteridium aquilinum, Picea spp, Rhododendron ponticum*

BOG: (CSM Indicators)
Positive: *Andromeda polifolia, Empetrum nigrum, Calluna vulgaris, Drosera spp, Erica spp, Sphagnum spp, Eriophorum spp, Vaccinium spp, Rubus chamaemorus*
Negative: *Agrostis stolonifera, Cirsium arvense, Holcus lanatus, Junus effusus, Pteridium aquilinum, Picea spp, Sorbus aucuparia*

[25] Bryophyte records have typically been omitted from analyses of CS data in the past. However, in a number of instances, as here,questions can be addressed by incoporating specific subsets of species data for which records can be considered reliable – see for example Bunce *et al* (1999b) and ROTAP (in press).

4.3 Rationale for selection

CSM Indicator richness is considered an informative yet transparent indicator of appropriate diversity primarily because it is based on existing species lists already agreed by the statutory agencies as indicators of habitat condition. CSM Indicators are also common enough, at least within their parent Broad Habitat, to support a count of indicators in plots as a useful variable. However, the rarer species more typical of pristine areas may be underrepresented in Countryside Survey plots. The data are therefore biased towards higher plants and toward the more common CSM Indicators. No weighting was applied to species to reflect differences in rarity across Britain or to reflect the membership of a species on multiple Priority Habitat lists.

As in the case of nectar plant diversity (Chapter 5), the rationale for the use and interpretation of the indicator is that it usefully conveys the varying richness of species in the wider species pools sampled by Countryside Survey. Therefore results should not be interpreted as an attempt to explicitly implement one facet of Common Standards Monitoring but as an attempt to provide quantitative contextual information on the diversity of CSM Indicators in the wider countryside. This is consistent with acknowledging that Countryside Survey is an unbiased sample of the unreserved matrix in which designated sites are embedded (Franklin 1993). CS also passively samples parts of designated sites but is not deliberately targeted on site interest features. Hence the indicator can be interpreted in two ways. Where Priority Habitats are likely to be extensive and so sampled by Countryside Survey plots, the indicator provides an unbiased assessment of the probable variation in richness of desirable or undesirable plant species across Priority Habitats. This interpretation is more reasonable in upland areas of Dwarf Shrub Heath and Bog. Where Priority Habitats are more fragmented and scarce the indicator is more likely to convey the potential of the local species pool to support or inhibit restoration and positive management depending on abundance and dispersal ability (Foster 2001; Critchley and Fowbert 2000; Bignal and McCracken 1996; Hodgson and Grime 1990).

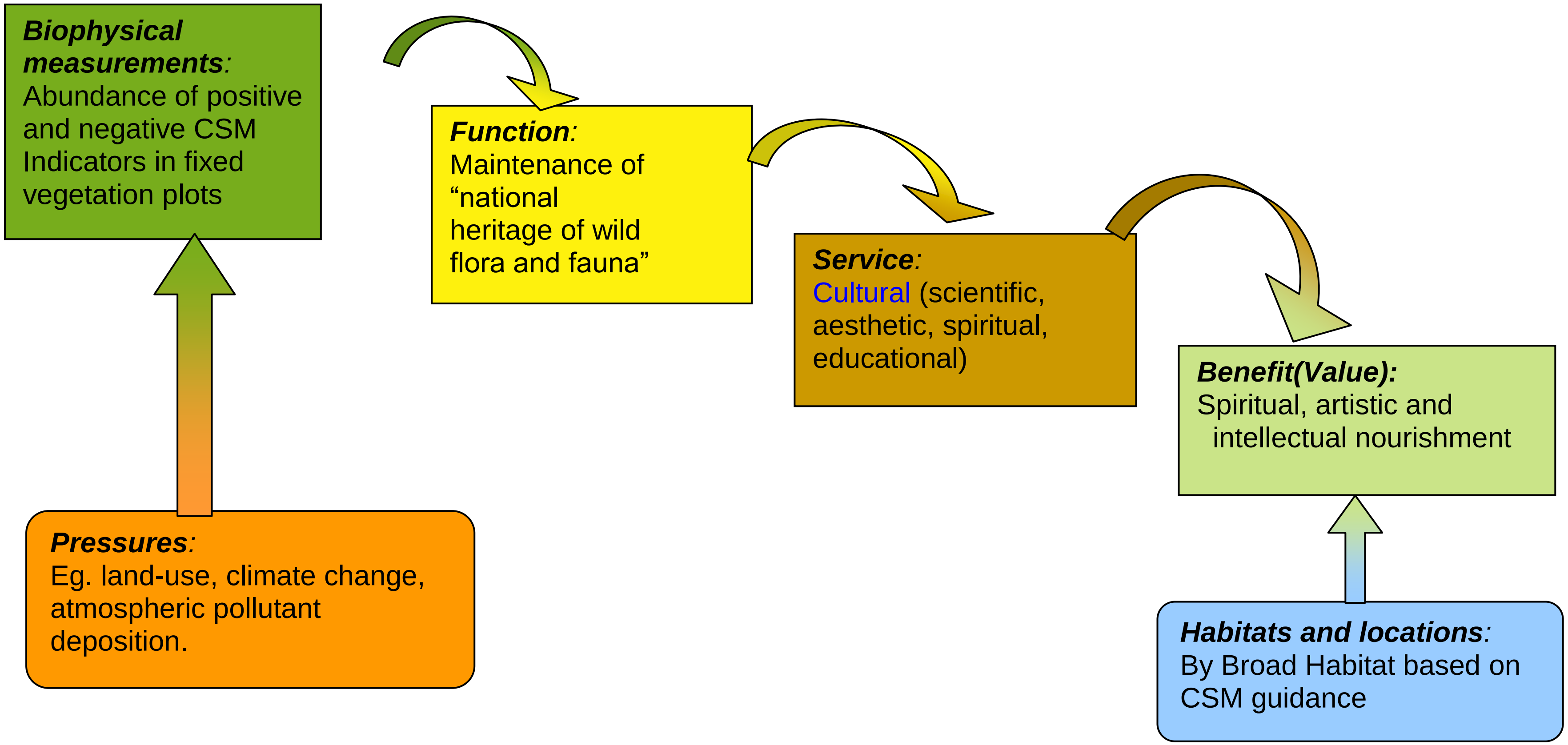
4.4 How is ‘appropriate diversity’ linked to ecosystem services?

‘Appropriate diversity’ targets a particular subset of the total pool of biodiversity across British ecosystems. By measuring abundance of positive and negative CSM Indicators (higher and lower plants) of condition for habitats of high conservation value as defined by the statutory conservation agencies, the indicator conveys potential delivery of the ecosystem service that is maximised when habitats are in ‘favourable condition’. Determining what category this service falls into requires close inspection of the motivations for designating sites of conservation value. CSM is applied to SSSI (Site of Special Scientific Interest) designated by the conservation agencies in accordance with a statutory duty under section 28 of the Wildlife and Countryside Act 1981 to notify any area of land which, in its opinion, is of “special interest” by reason of any of its flora and fauna. The Guidelines for Selection of Biological SSSI (Nature Conservancy Council 1989) describe how the designation process is the cornerstone of conservation practice in the UK where “The primary objective of nature conservation is to ensure that the national heritage of wild flora and fauna and geological and physiographic features remains as large and as diverse as possible, so that society may use and appreciate its value to the fullest extent.” Ten years after this document was published, CSM was made operational and currently provides a method for measuring condition on designated sites where condition relates explicitly to the interest features for which sites were designated. Attached to these features are conservation objectives which define what constitutes favourable condition of each feature in terms of a range of attributes and criteria. Criteria concerning presence or absence of characteristic species are based on the lists of CSM Indicator species that relate to each Priority Habitat and so provide a way of quantifying a key aspect of condition in a consistent manner across all designated examples of the Priority Habitat.

It seems clear that ‘favourable condition’ refers to the conservation of interest features of high conservation value rather than stewardship of the portfolio of supporting or regulating ecosystem services that a site may provide. In Britain, the designation of the SSSI and Natura 2000 series pre-dated notions of ecosystem service delivery. So there is a logical sense in which equating conservation objectives with ecosystem services must necessarily be a retrospective exercise. This recognises the principal aim of site designation was to conserve species and habitats for their own sake rather than as means to ensure supply of, for example, wild pollinators and nectar sources or to store carbon. Appropriate diversity is therefore best considered an aspect of cultural ecosystem service provision. It is cultural because conservation objectives reflect the value placed on what is special, scarce, natural and interesting in the British landscape; the “national heritage of wild flora and fauna” referred to above (Fig. 4.1). It is nevertheless easy to mistake this value-laden motivation for site designation as an objective exercise in conserving biodiversity-related ecosystem services. This is because the quantification of variation in biodiversity is heavily science-based and often emphasises the development and use of objective tools to ensure consistency and a common approach. However, the motivation for site designation is clearly to protect species and habitats for their own sake and as such their conservation fulfils a cultural demand sanctioned by society. The Guidelines for the Selection of Biological SSSIs (Nature Conservancy Council 1989) are again illuminating. Where the concept of “special scientific interest” is described (section 3.1) it is made clear that no guidance has ever been given in legislation as to its formal definition. Hence the statutory bodies have been left to decide on the conceptual framework and criteria for determination of “special interest” according to their “opinion”. The statement then follows that “The NCC has long understood biological interest to mean the wildlife values of an area to society for a broadly conceived range of cultural purposes, which include science, but also educational, recreational, aesthetic and inspirational values” (Fig. 4.1). On this basis the conceptual framework defining ecosystem services within the Millenium Ecosystem Assessment (MA 2003) would certainly regard the conservation of biological interest as a mechanism for delivering cultural ecosystem services.

As the quantification of biodiversity and the definition of interest features is science-based the outcomes of this process reflect the views of a well-educated elite who make decisions on behalf of society. We trust these judgements based on assumptions about their expertise and experience, which from time to time have been tested in the courts and at public inquiry. However, because the site designation and monitoring process is suffused with the scientific method it is again easy to conflate the conservation of species and habitats with objective stewardship of ecosystem functions and ecosystem service provision. Again it should be emphasised that CSM Indicators are first and foremost indicators of conservation value. The fact that achieving this in any one habitat may bring with it soil carbon storage, water flow regulation and high pollinator diversity (see Chapter 7) does not make the term any the less principally a cultural criterion. For example, if the regulating or supporting ecosystem services provided by a habitat could be delivered via technology this would not make it more likely that the destruction of the habitat or its deterioration was sanctioned by the statutory conservation body; the conservation of the habitat for its own sake and with optimal appropriate diversity would still remain as the principal objective.

Figure 4.1: The ecosystem service cascade for 'appropriate diversity' (after Haines-Young and Potschin 2007).



4.5 Current status and trends across GB

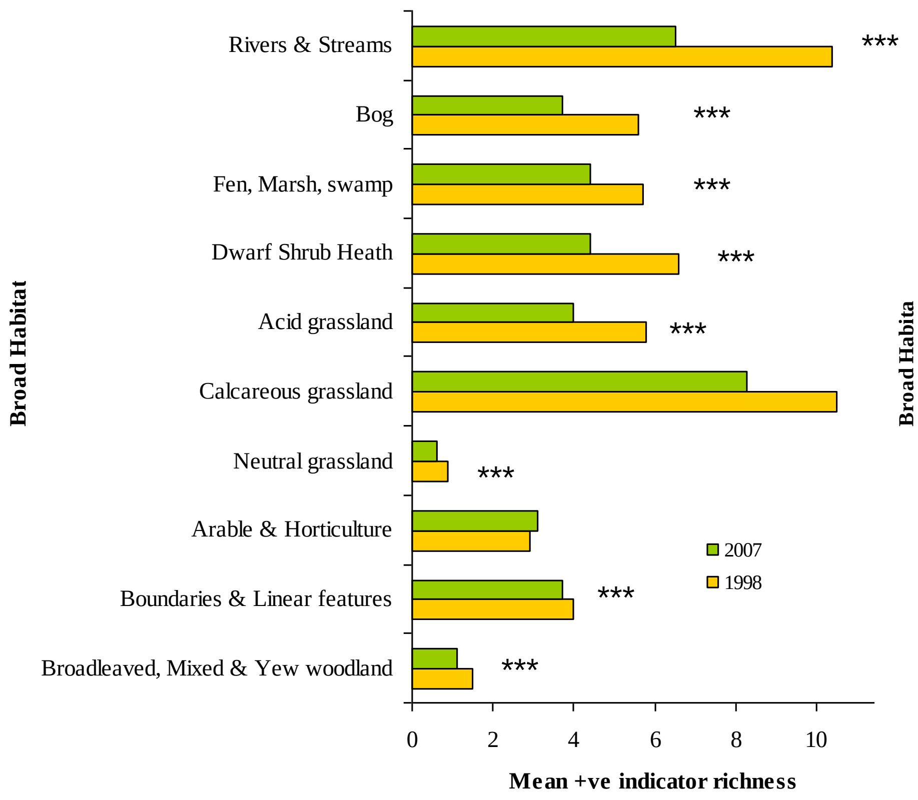
Between 1998 and 2007, the mean richness of positive CSM Indicators declined significantly in all Broad Habitats analysed except in the Arable Field Margin Plots in the Arable and Horticulture Broad Habitat (Fig. 4.2a). This is consistent with the general decline in mean species diversity observed in Countryside Survey in this interval. This change accompanied a GB-wide signal of reduced disturbance and fewer vegetation gaps, a pattern also seen between 1990 and 1998 (Carey *et al.* 2008) and one also seen in the independent series of Environmental Change Network sites between 1994 and 2007 (Moorcroft *et al.* 2009). The largest declines in positive indicator richness were in Calcareous Grassland (not significant probably because of small sample size) and Rivers and Streams. Both habitats had the highest starting richness and hence had more species to lose. In the Arable and Horticulture Broad Habitat a small non-significant increase was seen yet the magnitude of the change was much smaller than the 30% increase in total species diversity in Main Plots located away from the margin reported for CS in 2007 (Carey *et al.* 2008). This is probably because A Plots are always constrained to sample the cultivated edge of the crop. Hence, if surveyors judge that the field margin is no longer under cultivation then the A Plot location moves into the field to sample the new cultivated crop margin. In addition the large size of the A Plots mean that most dicot species may already have been present even if at low abundance thus reducing scope for further increases in richness[26].

Decreased mean richness of negative indicators was also seen across all the Broad Habitats analysed and these changes were significant in all but three Broad Habitats (Fig. 4.2b). Directions of change in mean cover of negative indicator species varied between Broad Habitats. Cover increased in the Boundaries and Linear Features and in the Rivers and Streams Broad Habitats consistent with successional signals (Fig. 4.3). Cover also increased significantly in Acid Grassland but decreased significantly in Neutral Grassland and Bog (Fig. 4.3).

[26] The series of M Plots (1x1m on a transect into the field) newly placed in CS squares in 2007, were specifically designed to be a baseline for assessment of future changes in field margins under new agri-environment scheme prescriptions. See Carey *et al* (2008) for further details.

Figure 4.2: Mean richness of a) positive and b) negative indicators of 'appropriate diversity' in Britain in 2007 and 1998. Richness of arable dicots is presented for A Plots (1x100m) located along field boundaries in the Arable and Horticultural Broad Habitat. Ancient Woodland Indicator (AWI) richness is shown for plots in the Broadleaved Woodland Broad Habitat. In other Broad Habitats richness of positive Common Standards Monitoring (CSM) Indicators is shown. CSM Indicators were taken from the JNCC guidance for associated Priority Habitat .

a)



b)

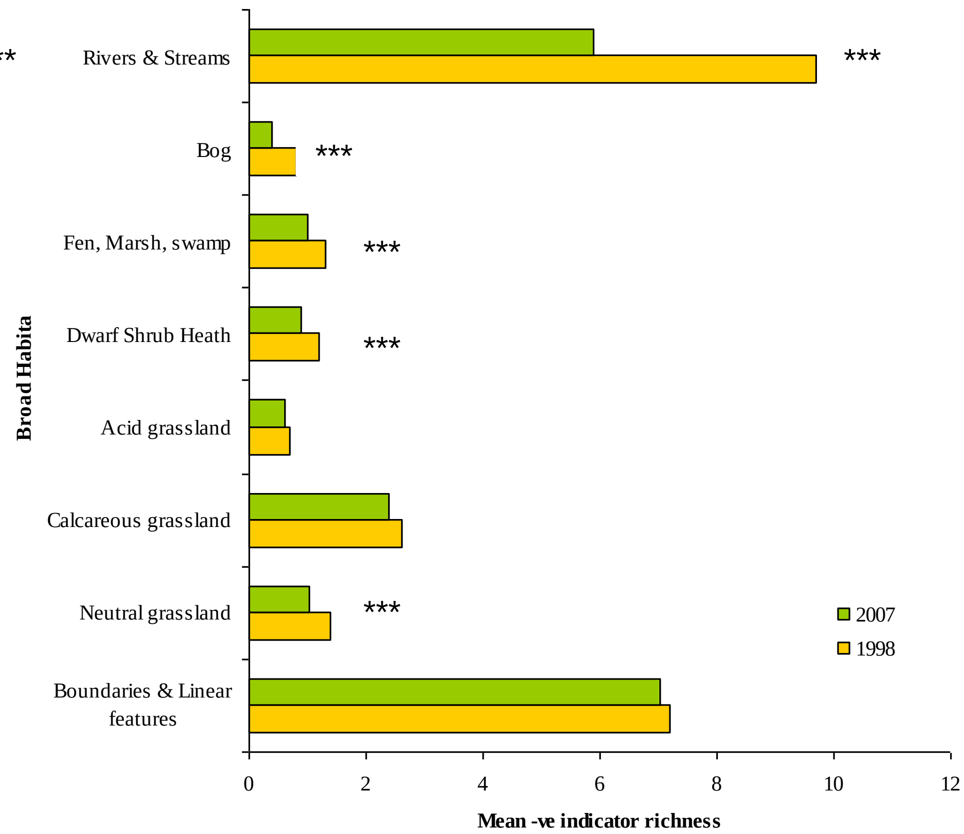
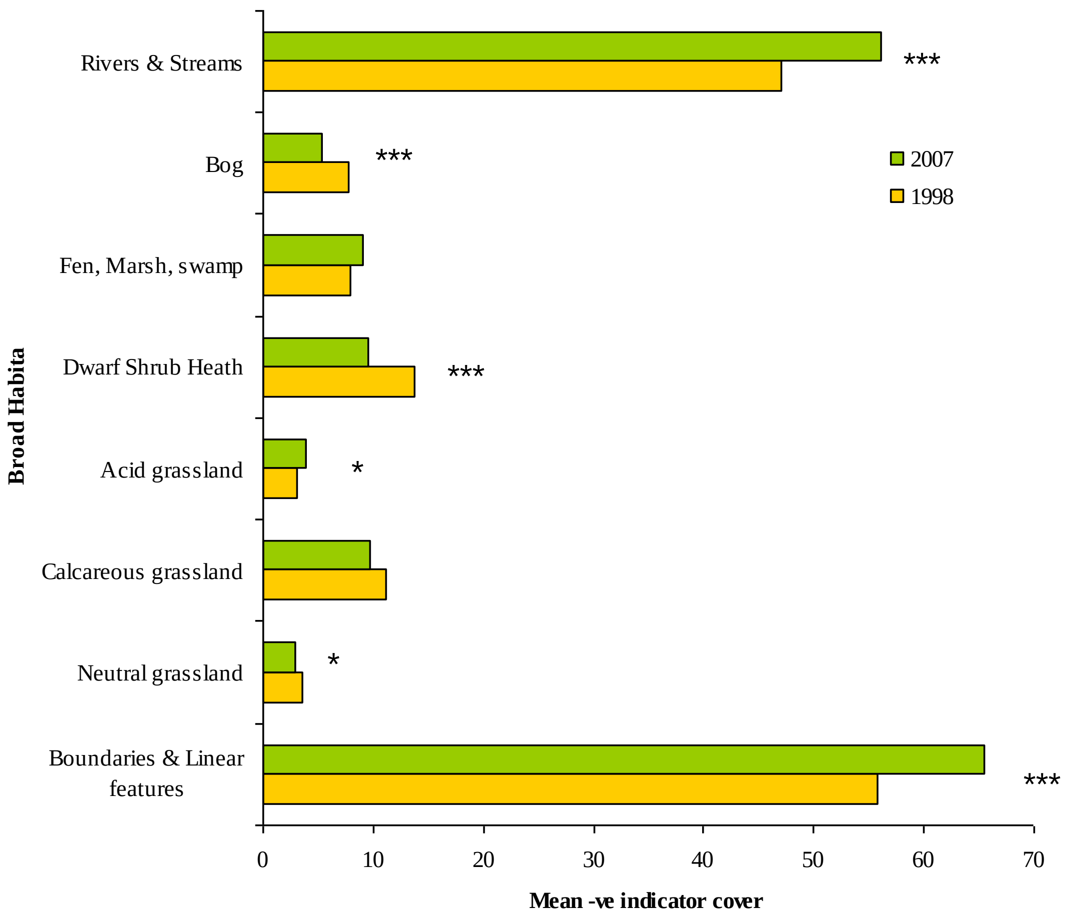


Figure 4.3: Mean cover of negative Common Standards Monitoring (CSM) Indicators in Britain in 2007 and 1998.



4.6 Maps of status and change

Is it possible to meaningfully map ‘appropriate diversity’?

The main constraint on mapping appropriate diversity is having sufficient data to represent spatial variation in abundance of CSM Indicators across the wider landscape. CS provides counts of CSM Indicator richness from small vegetation plots in a sample of British 1km squares (591 in 2007). Although an unbiased sample, only a small fraction of the total area of designated sites is represented. Hence, rarer CSM Indicators and rare habitat types will be under-represented and more common CSM Indicators and habitat types over-represented compared to the picture gained from condition assessment carried out across British SSSI (Williams 2006). Therefore CS data cannot in anyway be used or interpreted as a repeat or a validation of CSM. Rather, the results give an unbiased contextual picture of the abundance and change in abundance of CSM Indicators in the wider countryside inside and outside designated sites and across the landscape irrespective of the fit of each sampled patch to any one Priority Habitat. Mapping CSM Indicator species richness across Broad Habitats across Britain is only likely to be useful if this wider, unbiased yet general, picture is deemed useful. Even if it is possible to map average abundance of these species, the number of listed CSM Indicators differs between Priority Habitats and between Broad Habitats. A simple map of absolute richness values will simply reflect differences in the length of each CSM Indicator list, obscuring any ecologically interesting spatial patterns. As well as CSM count differing simply as a function of the length of each Priority Habitat list, ecological factors are also likley to be important in explaining spatial variation. Counts in plots could vary with geographical location in addition to site management status, patch size and other factors. Hence, the simplest option, mapping GB-wide Broad Habitat average richness wherever the Broad Habitat occurs, is less informative because this takes no account of geographical variation or any of the factors that drive this spatial variation. This lack of spatial sensitivity would still apply even if we attempted to express richness as a proportion of the total indicator list for each Broad Habitat.

Given sufficient time, spatial regression models could potentially be constructed (see for example Chapter 5). The simplest unified model could include Broad Habitat as the main explanatory variable with additional effects of habitat patch size, total habitat extent in the sampled region, pollutant deposition, climatic variables and intensive land-use extent. These models might be informative in quantifying broad landscape-scale constraints on the richness of each Broad Habitat-based species pool. In the absence of such Broad Habitat-specific models another option is to quantify variation in the total abundance of all CSM Indicators across the landscape but comparing the species pool in larger areas of common habitat with the diversity of the species pool in residual fragments of semi-natural habitat and in linear features. Many of the semi-natural Broad Habitats, such as Neutral

Grassland, Fen, Marsh and Swamp, Bog and Dwarf Shrub Heath, cover extensive areas of upland Britain but are fragmented and scarce in lowlands.

Previous work has shown that small habitat fragments and linear features can support higher residual diversity of desirable plant species than surrounding production lands (Smart *et al.* 2006). The same question could be posed for CSM Indicators generally; do linear features and small habitat fragments differ predictably in richness of positive or negative CSM Indicators from adjacent larger areas of common habitats and unenclosed land across upland and lowland Britain? The expectation would be that at very low land-use intensity both linear features and surrounding areas realise their optimum positive indicator richness but, when not intensively managed, common areal habitats are likely to be richer because they are more optimal (Smart *et al.* 2006). Moving along the land-use intensity gradient, and therefore from upland to lowland land classes, linear features and small habitat fragments ought to become relatively richer but species richness in all landscape locations is expected to fall. Maps of land class means ought to allow a test of these expectations but any variation within the land class will be averaged out. This will lead to a less noisy but less spatially detailed visualisation since an unknown amount of meaningful variation at the sub-land class level will not be expressed. To explore whether land class maps could convey useful national patterns and hence provide a simple option for mapping an indicator of appropriate diversity, ITE Land Class maps were constructed showing how the difference in total CSM Indicator richness between potential refuge features and adjacent areas varied across Britain. Results are presented and discussed in Appendix 4.2.

4.7 Explaining change in 'appropriate diversity' between 1998 and 2007

Evidence for impacts of key drivers on Common Standards Monitoring (CSM) Indicator species abundance

CSM Indicator species as a classification of taxa, only date back to 1999 when CSM became operational across UK designated sites (Williams 2006). However, insights into the likely drivers of change in appropriate diversity in these species groups can be readily gained from background information on changes in habitat extent and condition that pre-date CSM guidance. The interval covered by the attribution analysis also coincides with UK Biodiversity Action Plan progress reporting and with the first six year report on SSSI condition (Williams 2006). Both sources provide information on adverse activities that highlight specific drivers of change in condition as reported across Britain for 2005. Pressures were also linked to Broad Habitats in Haines-Young and Potschin (2007) reflecting the views of the lead agencies for relevant parts of the UK BAP. All these sources often rely on the apparent parallel declines in biodiversity and increase in pressures but do not offer direct quantitative evidence of a relationship between change (spatial or temporal) in a driving variable and correlated ecological impacts and are seldom able to quantify the relative importance of different pressures, for example atmospheric nitrogen deposition versus over-grazing, given that the two often act together.

Evidence from wider-countryside surveillance; non-random effects on plant species

Consistent messages regarding the kind of plants that have been 'winners' and 'losers' across the countryside emerge from Countryside Survey (Carey *et al.* 2008; Smart *et al.* 2005), Local Change (Braithwaite *et al.* 2006), Atlas 2000 (Preston *et al.* 2000) and a number of other regional and national analyses (Hodgson *et al.* 2005; Kirby *et al.* 2005; Walker 2003; Walker *et al.* 2009). Albeit with important variation between upland and lowland habitats and between habitat types (Hodgson *et al.* 2005), winning plants have tended to be tall, nutrient-demanding and often graminoids or woody species. Losers have tended to be short, stress-tolerant forbs. Given that these trait syndromes could be used to broadly differentiate positive from negative CSM Indicators (Fig. 4.4), it is reasonable to expect that drivers of this non-random filtering of the species pool in the wider-countryside could also be correlated with increases and decreases in CSM Indicator groups.

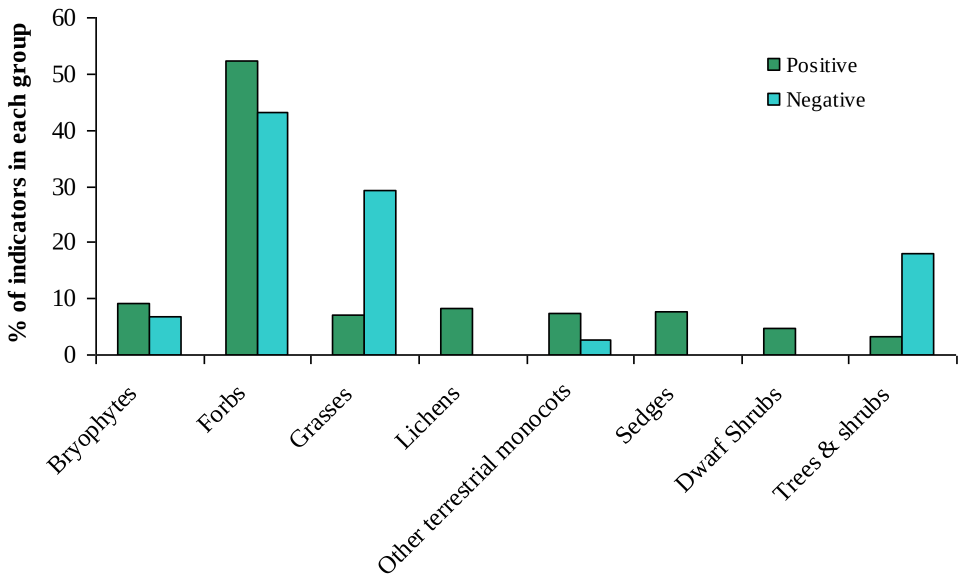
Impacts of positive management

Our approach to selection of drivers and explanatory variables was guided by expert advice and a literature search, but was also constrained by the availability and resolution of datasets. A substantial gap relates to spatially comprehensive yet finely resolved data on the coincidence of Countryside Survey sample plots and positive management schemes. Selection of explanatory variables largely reflected hypotheses about negative impacts of drivers on appropriate diversity, yet in 2006 41% of the agricultural land area of the UK was reported to be in some form of agri-environmental scheme and 44% of woodland was under certified sustainable management[27]. Hypotheses of the effects of positive management were included where datasets were available that tracked the implementation of such policies at useful scales. However, having gone to considerable lengths to assemble attribution datasets, it is notable that highly resolved data on the location, history and details of positive management are much more scarce than information on the drivers that are likely to degrade condition. This very likely results in a bias in the results since improvement and wider maintenance will not be attributed to the policies and schemes that might be responsible. Whilst digital spatial layers are readily available for some countries for scheme coverage they are not available in others. In addition, simple coincidence with designated areas such as SSSI and Environmentally Sensitive Areas (ESA), do not provide

[27] Defra (2007) Biodiversity Indicators in Your Pocket. http://www.jncc.gov.uk/pdf/2010-BIYP2007.pdf. These high figures emphasise the problem. The actual area of land impacted directly by a change in management is likely to be very much lower; possibly below 10% of agricultural area (J.Hopkins, pers.comm.) but precise data at the polygon or sub-polygon level that conveys the location, history and extent of direct impacts is largely unavailable.

enough detail on polygon level management history to enable effective selection of data subsets comprising unimpacted areas and areas known to have been managed for long enough to produce a likely response with both datasets having a common vegetation starting point (Critchley *et al.* 1996).

Figure 4.4: Plant growth forms represented among positive and negative Common Standards Monitoring (CSM) Indicators recorded in Countryside Survey.



Hypothesising links between change in 'appropriate diversity' and potential drivers of change

Sources of evidence described in the previous sections were used to create hypotheses about expected correlations between change in appropriate diversity and possible drivers. The selection of explanatory variables varied somewhat between location and habitat type (Tables 4.1 and 4.2). Hypotheses were arranged by Broad Habitat reflecting the nesting within the Broad Habitat, of those Priority Habitats which have been furnished with CSM Indicator species.

Two explanatory variables (f and g in Table 4.2) were constructed to quantify the impact of a change along substrate productivity or succession/disturbance axes. The approach was to measure change in the plant species composition as the difference between ordination axis scores in each plot between 1990 and 1998. These scores were derived from an overall ordination of CS data that has shown how floristic variation is organised along these two principal axes (Bunce *et al.* 1999a; Smart *et al.* 2003). Thus, differences in scores between surveys on axis 1 and 2 indicate changes along a species-compositional gradient associated with fertility or successions/disturbance respectively. These differences in scores between 1990 and 1998 were used as explanatory variables for change in appropriate diversity between 1998 and 2007. This approach was used for two reasons. First, circularity is avoided; although the 1998 to 2007 change maybe correlated with change between 1990 and 1998, the two aren’t measuring aspects of the same species assemblages at the same points in time. Secondly, this approach is consistent with a legacy or conditioning effect of previous change in resource availability that is expected to make a patch susceptible to invasion by new species. This could increase or decrease species richness depending on abiotic starting point and the traits of the species lost or gained (Davis *et al.* 2005; Wright and Jones 2004). The impact of inferred changes in ecological conditions on subsequent change in appropriate diversity differ depending on whether negative or positive indicators are concerned and with variation in the starting abiotic conditions in 1990 (see Table 4.2). Data sources for other explanatory variables in Table 4.2 are described in Box 4.2.

Sources of explanatory variables — *Box 4.2*

a. HUMAN POPULATION DENSITY: Based on data for 2007 from the Office of National Statistics and General Register Office of Scotland.

b & c. NOy AND NH_X DEPOSITION: Estimates of oxidised and reduced nitrogen were provided by CEH Edinburgh at 5km x 5km scale. The data are interpolated 3-year means for 2004-’06 based on UK monitoring networks and a new methodology described in Fowler *et al* (in press). Data were provided as estimates of deposition to forest, moor, improved grassland and arable and values attached to each set of Broad Habitat-based plots as appropriate. Deposition variables were not used for analysis of Arable & Horticulture given that crops typically emit nitrogen and fertiliser inputs greatly exceed any ambient deposition.

d & e. CHANGE IN RAINFALL AND TEMPERATURE: Change was calculated on centred and standardised data as the linear slope coefficient of annual 5km x 5km rainfall and mean temperature data from 1980, the estimated start of the recent warming trend across the UK, up to 2003, the latest year for which climate data could be obtained. All data were interpolated estimates downloaded from the Met Office data store.

f & g. CHANGE ALONG PRINCIPAL ECOLOGICAL GRADIENTS; 1990 TO ’98: See main text.

h & i. SHEEP DENSITY CHANGE: Estimates at 2km x 2km scale were obtained from the EDINA AgCENSUS database. Changes in sheep density per grid square were calculated as the linear slope coefficient from 1969 to 2000 just prior to foot & mouth (h) and then separately from 2000 to 2004 to track regional post foot & mouth reductions.

Table 4.1: Hypothesised drivers of change in 'appropriate diversity'. The top three adverse activities reported for each habitat as part of CSM monitoring on designated sites between 1999 and 2005 are reported separately for upland (u) and lowland (l) except for Bogs, which are divided into Blanket Bog (b) and Lowland Raised Bog (r)[28]. See Table 4.2 for codes to explanatory variables tested for each habitat type.

	Broadleaved, Mixed and Yew Woodland	Neutral Grassland	Arable and Horticulture	Acid Grassland	Calcareous Grassland	Fen, Marsh and Swamp	Bog	Dwarf Shrub Heath	Rivers and Streams (banksides)	Boundary and Linear Features
Hypothesised gradients	Shading, eutrophication	Extensification and recovery	Extensification and recovery	Stability or local succession post foot and mouth	Succession and eutrophication	Succession and eutrophication	Stability or eutrophication	Recovery post foot andmouth or stability plus eutrophication	Shading and eutrophication	Shading and eutrophication
Explanatory variables – see table 4.2	a, b, c, d, e, f, g	a, b, c, d, e, f, g	a, d, e, f, g	a, b, c, d, e, f, g, h, i	a, b, c, d, e, h, i	a, b, c, d, e, f, g, h, i	a, b, c, d, e, f, g, h, i	a, b, c, d, e, f, g, h, i	a, b, c, d, e, f, g	a, b, c, d, e, f, g
Top three 'adverse activities' (CSM 2006)	1, Forestry 2, Overgrazing 3, Invasive species	1, Under-grazing 2, Agricultural operations 3, Lack of remedial management	N/A	1u,Over-grazing, 2u, 1l, Under-grazing 3u. Agricultural operations 2l, Lack of remedial management 3l, Invasive species	1l, Under-grazing 2l, Lack of remedial management 3l, 2u, Invasive species 1u, Over-grazing 3u, Agricultural operations	1u, Overgrazing 2u, Agricultural operations 3u, 3l, Water management 1l Lack of remedial management 2l, Under-grazing	1b, Over-grazing 2b, Burning 3b, 1r, Water management 2r, Lack of remedial management 3r, Invasive species	1u,Over-grazing, 2u, Burning 3u, 1l, Lack of remedial management 2l, Under-grazing, 3l, Invasive species	1, Water quality 2, Water management 3, Overgrazing	N/A

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28 Notes on most important 'adverse activities' indicated in CSM 2006. A) CS patches may not be representative of the larger, better quality SSSI areas and this could explain differences in severity and identity of drivers. B) In the case of Neutral Grassland, SSSIs are more likely to target scarce hay meadow Priority Habitats while CS includes a wider range of much more common semi-improved grasslands (MG6) as well as the less common community types (see Jackson 2000). Hence SSSI unimproved meadows maybe more likely to be undergrazed than Neutral Grassland represented in CS where much of the mapped extent would still be widely exploited for sheep and cattle grazing, hay and silage. C) For Fen, Marsh and Swamp, the possibility that recent declines in cattle and sheep numbers across Britain have caused a successional response since 1998 can still be consistent with areas being considered to still be in unfavourable condition because they are overgrazed. For example, lower grazing pressure could trigger expansion of –ve indicator species as well as +ve but if previous overgrazing has depressed the abundance of the latter then this could justify assignment as unfavourable even if grazing has been recently reduced.

Table 4.2: Expected relationships between explanatory variables and change in abundance of indicators of ‘appropriate diversity’.

	Positive indicator richness	Negative indicator richness	Negative indicator cover
a. Population density	-ve impact because of greater likelihood of inappropriate disturbance, dispersal of competitive non-native and native species, smaller areas of habitat in lowlands and more likely to be unmanaged (Hodgson *et al.* 2005), +ve correlation for some habitats that are more species rich in southern GB where population density is also highest.	+ve effect because of lack of dispersal limitation and exposure to nutrient surpluses	+ve effect because of pulse disturbance effects and exposure to nutrient surpluses
b. NOy deposition	Eutrophication ought to cause a +ve effect for habitats to left of modal species richness and –ve for those at the mode and to the right (Smart *et al.* 2003). Recent temporal correlation expected to be small or not significant because historical effects already played out (Maskell *et al.* 2010; Stevens *et al.* 2006).	+ve effect because of eutrophication	+ve effect because of eutrophication
c. NHx deposition	Larger -ve effect because of the impacts of acidification and eutrophication but comments above also apply.	+ve effect because of eutrophication but recent temporal correlation expected to be small or not significant because historical effects already played out (Maskell *et al.* 2010; Stevens *et al.* 2006).	+ve effect because of eutrophication but recent temporal correlation expected to be small or not significant because historical effects already played out (Maskell *et al.* 2010; Stevens *et al.* 2006).
d. Change in annual rainfall 1980 to 2003	Uncertain; if lower rainfall and warmer summers create gaps and reduce vigour of dominants then +ve effect expected (Morecroft *et al.* 2004; 2002) but depends on habitat type ie. +ve CSM that depend on high summer water table would suffer while upland habitats can show considerable resistance (eg. Grime *et al.* 2008).	Uncertain; comments for positive indicators apply here as well.	Wetter summers and warmer winters should promote herbaceous dominants (Dunnett *et al.* 1998) including grasses (Silvertown *et al.* 1994; Morecroft *et al.* 2004) but context dependent since very high water table can eliminate wet and dry grassland species (Critchley *et al.* 1996) while increased tree and shrub growth could suppress herbaceous competitors (Smart *et al.* 2006a).

e. Change in mean annual temperature 1980 to 2003	-ve effect by promoting competitive advantage of negative indicators but resistance and lack of change possible in uplands despite greatest increases in growing season length (Grime *et al.* 2008)	+ve effect if rainfallalso increasese with no summer drought (Dunnett *et al.* 1998); comments for positive indicators apply here as well (Grime *et al.* 2008).	+ve effect if rainfallalso increasese with no summer drought (Dunnett *et al.* 1998); comments for positive indicators apply here as well (Grime *et al.* 2008).
f. Change toward more fertile conditions 1990 to ‘98	-ve effect if legacy is felt through to 2007 because a change to more productive conditions will not favour positive indicators	+ve effect if legacy is felt through to 2007 because a change to more productive conditions is likely to favour negative indicators	+ve effect because a change to more productive conditions is likely to favour negative indicator cover
g. Change to less disturbed conditions 1990 to ‘98	Dependent on context; over-grazed Acid Grassland might benefit but an increase in sward height would not favour lichens and small herbs, especially annuals. Recovering heathland dwarf shrubs would benefit. Less disturbance in Fen Marsh and Swamp, Dwarf Shrub Heath and woodland might also have a negative impact but positive responses could be expected in less intensively managed Blanket Bog. +ve effect possibly expected in Arable plots but note that only total arable dicot richness was analysed since no CSM list exists for arable field margins. Imapct of extensification expected 2-4 years after removal from cultivation so a signal might be detectable (Critchley *et al.* 2006; Muster *et al.* 2008). However, since A Plots targeted the cultivated crop edge they may not capture new perennial margins.	+ve effect on most species except in late-successional habitats eg. woodland, boundaries, wooded streamside where greater shade would be likely to reduce vigour of positive as well as negative shade-intolerant herbs (Smart *et al.* 2006a).	As for negative indicator richness
h.Sheep change 1969-2000	-ve effect in upland habitat types since the period coincides with the major increase in sheep numbers through the late 70s and 80s driven by CAP headage payment incentives (Fuller and Gough 1999).	+ve effect since many of the negative indicators are clonal grasses and grazing tolerant herbs encouraged by grazing and decreases in surface soil C:N (Pakeman *et al.* 2004; Smart *et al.* 2007; Hodgson *et al.* 2005).	As for negative indicator richness
i. Sheep change 2000-2004	While localised in the British uplands, the decline in the national flock post-foot and mouth may have been enough to drive increases in species density of positive indicators.	No change expected since reduced grazing intensity would be likely to encourage medium term growth of the negative indicators.	As for negative indicator richness

Analytical methods

Analysis focused on testing the significance of change conditional on each explanatory variable between 1998 and 2007. Plots were grouped by the Broad Habitat in which they were located in 2007.

Statistical analysis employed Generalised Linear Mixed Models written in the SAS environment (Little *et al.* 2000). The approach is fully described in Maskell *et al.* (2010). In essence, the 1 km CS squares, within which plots are nested, were treated as random variables drawn from a normal distribution. This ensures that the correct test statistics are calculated given the distribution of the variability between and within squares. Also, the degrees of freedom were downweighted based on the similarity of the responses within each square. This guards against false significance resulting from the treatment of all plots as statistically independent.

Hypothesis tests of the existence of correlations between explanatory variables and change in indicator richness or cover were carried out by testing the fit of each explanatory variable to the sample of differences in richness or cover having first fitted all other explanatory variables as covariates (see Table 4.2). Using this approach, attribution of change in appropriate diversity to each explanatory variable relies on there having been a long and steep enough gradient in the hypothesised driver to have generated the statistical power for detection of a correlated ecological impact. If a driver had operated with equal severity everywhere and an associated magnitude of impact had occurred everywhere then the only deviations from the mean of the response will be random about the mean overall level of the explanatory variable effectively ruling out the possibility of a systematic association between the two and hence a significant detected effect of the driver on the response. This means that the strongest attribution will arise where areas with an absence of the driver are interspersed with areas with the highest levels of operation of the driver along a gradient of well replicated values in between.

Results

Out of 38 significant relationships between explanatory variables and change in indicator abundance, 14 were consistent with expectation, 13 were inconsistent and 11 were not consistent with the most obvious mechanism but could be consistent with plausible alternative explanations. No significant relationships between explanatory variables and change in indicator abundance were detected for Fen, Marsh and Swamp. Montane and coastal Broad Habitats were not analysed because they are poorly represented in CS data.

Significant drivers of change in 'appropriate diversity'

The most frequent significant relationships between change in indicator abundance and explanatory variables involved: a) pre-conditioning changes in species composition between 1990 and 1998 along axes of fertility and succession, b) change in mean annual temperature, and c) nitrogen deposition. In no case was change in sheep density between 2000 and 2004 significant.

Correlations consistent with expectation

Consistent correlations with modelled atmospheric nitrogen deposition were seen in Boundary and Linear Features, where positive CSM Indicators were more likely to decline at higher deposition values, while in Neutral Grassland and Dwarf Shrub Heath higher nitrogen deposition was associated with a greater chance of increasing richness of negative CSM Indicators, and greater chance of increased negative indicator cover in Boundary and Linear Features, Dwarf Shrub Heath and Acid Grassland (Table 4.3).

A significant negative correlation between change in sheep numbers between 1969 and 2000 and change in richness of positive CSM Indicators was seen in the small Calcareous Grassland sample. The results suggests that increases in positive indicator richness are less likely and decreases more likely in areas subject to historically higher increases in sheep density. The mechanism of this legacy effect and its location requires closer inspection although the pattern is consistent with the negative impact of overgrazing (Table 4.3).

Other results that were consistent with expectation include the negative impact of succession on richness of positive CSM Indicators on river and stream banks and also the interesting positive correlations between change in mean annual temperature and cover of negative CSM Indicators in Neutral Grasslands, Boundary and Linear Features and Rivers and Streams (Table 4.3). These results detect, for the first time, a significant signal of climate change impact on the abundance of species that influence condition assessment on designated sites. The positive correlation with negative indicators is consistent with the expected impact of warmer and wetter conditions on more nutrient-demanding species (Dunnett *et al.* 1998) a climatic trend that appears to have occurred across Britain since at least the early nineties (Moorcroft *et al.* 2009).

Inconsistent correlations

Assuming that positive indicators tend to be more stress-tolerant and less able to tolerate or exert competitive effects above-ground, the positive associations between mean annual temperature change and positive indicator richness change in Broadleaved, Mixed and Yew Woodland, Neutral Grasslands and Boundary and Linear Features are unexpected as is the positive correlation between mean annual rainfall change and positive indicator richness change in Boundary and Linear Features (Table 4.3). The negative correlation between negative indicator richness and surrounding human population density for Calcareous Grassland was also in the opposite direction to that expected.

A number of inconsistent correlations were detected between changes in indicator richness between 1998 and 2007 and movement along the ordination axis related to substrate productivity between 1990 and 1998. These were seen in Arable and Horticulture, Rivers and Streams, Boundary and Linear Features, Neutral Grassland, Acid Grassland, Dwarf Shrub Heath and Bog. Legacy effects of previous change to a more productive species composition between 1990 and 1998 were correlated with increases in positive CSM Indicators in Rivers and Streams and Boundary and Linear Features and in arable dicots in Arable and Horticulture. Conversely, the same legacy effect was correlated with a decrease in richness of negative CSM Indicators in Bog.

Lastly, four significant negative correlations between change in negative CSM Indicator cover and oxidised nitrogen deposition were detected in Neutral Grassland, Acid Grassland, Boundary and Linear Features and Rivers and Streams. These correlations run counter to the known fertilising effect of oxidised forms of nitrogen.

Correlations that could be consistent with alternative mechanisms

The positive correlation between change in richness of Ancient Woodland Indicators and change to a later successional species assemblage between 1990 and 1998 could be understood if the increasing indicators were shade-tolerant species. However, most of these species exploit woodland gaps and mean richness of this group declined alongside mean richness of all species in the Native Woodland Survey, a change strongly linked to increased canopy shade and lack of managed disturbance (Kirby *et al.* 2005).

The positive correlation between change in positive indicator richness and reduced nitrogen deposition in Dwarf Shrub Heath could be considered consistent with a cross-species fertilising effect. Indeed the habitat type is on the left of the unimodal species diversity-productivity gradient (Smart *et al.* 2003) which would predict a general increase in diversity with increasing productivity. However, the positive effect on positive indicators is at odds with work identifying indicators of reduced nitrogen impacts in upland vegetation in Britain (Smart *et al.* 2007).

Other correlations that seem inconsistent with the most obvious expected direction of change were all negative correlations between change in cover or richness of negative CSM Indicators and the legacy effect of change toward later successional or more productive vegetation between the earlier surveys in 1990 and 1998. The implication is that a shift toward conditions that ought to favour negative CSM Indicators has actually had the opposite effect. A possible explanation is that successional change has resulted in sufficient tree or shrub dominance to reduce the vigour of the negative indicators. This is consistent both with the known increase in woody cover on British streamsides (Carey *et al.* 2008) as well as the potentially suppressive effect of woody cover on nutrient-demanding and shade-intolerant herbs that feature prominently in the negative indicator list (eg. Smart *et al.* 2006; Kirby *et al.* 2005).

Table 4.3: Attribution evidence for effects on 'appropriate diversity' by Broad Habitat in Countryside Survey between 1998 and 2007.

	Broadleaved, Mixed and Yew Woodland	Neutral Grassland	Arable and Horticulture	Acid Grassland	Calcareous Grassland	Fen, Marsh and Swamp	Bog	Dwarf Shrub Heath	Rivers and Streams (banksides)	Boundary and Linear Features
+ve indicator richness change	Positive correlation with climate warming since 1980 Positive correlation with successional change	Positive correlation with climate warming since 1980	Positive correlation with reduced disturbance Positive correlation with eutrophication signal ('90-'98)	Ns	Negative correlation with increase in sheep numbers up to 2000	Ns	Ns	Positive correlation with reduced N deposition	Positive correlation with eutrophication signal ('90-'98) Negative correlation with successional change	Negative correlation with reduced N deposition Positive correlation with climate warming since 1980 Positive correlation with rainfall change since 1980 Negative correlation with eutrophication signal ('90-'98)
-ve indicator richness change	n/a	Positive correlation with oxidised N deposition Negative correlation with successional change	n/a	Negative correlation with eutrophication signal ('90-'98)	Negative correlation with population density	Ns	Negative correlation with eutrophication signal ('90-'98)	Positive correlation with reduced N deposition	Negative correlation with reduced N deposition	Negative correlation with reduced N deposition Negative correlation with oxidised N deposition

										Positive correlation with climate warming since 1980 Negative correlation with eutrophication signal ('90-'98) Negative correlation with successional change
-ve indicator cover change	n/a	Negative correlation with oxidised N deposition Positive correlation with climate warming since 1980 Negative correlation with successional change	n/a	Positive correlation with reduced N deposition Negative correlation with oxidised N deposition	Negative correlation with successional change	Ns	Ns	Positive correlation with reduced N deposition Negative correlation with oxidised N deposition	Negative correlation with successional change Positive correlation with climate warming since 1980	Positive correlation with reduced N deposition Negative correlation with oxidised N deposition Positive correlation with climate warming since 1980

4.8 Discussion

Development of an indicator of 'appropriate diversity'

In this chapter we have argued that conservation value primarily reflects a cultural ecosystem service despite the dividend that site safeguard and conservation management may bring in terms of securing supply of other services. In the wider countryside sampled by CS, a transparent indicator for measuring status and change of aspects of this cultural service can be based on the diversity of indicator species listed for particular habitats of high conservation value. This is a measure of appropriate diversity because the species composition of each list is appropriate to each BAP Priority Habitat. In combination with the novel work presented in Chapter 6, this analysis therefore contributes a new and simple practical method and metric for measuring cultural service status in the wider countryside.

In incorporating published CSM Indicator lists into a method for use across CS sample squares it is important to emphasise that this is not an attempt to repeat Common Standards Monitoring outside designated sites. The approach, as with many applications of CS data, is to uniquely and informatively quantify the spatial and temporal ecological context within which designated sites and other lands under conservation management agreements are embedded (e.g. Carey *et al.* 2002). A further related caveat on the use of CS data to enumerate CSM Indicators in the wider countryside is that species coverage will also differ significantly from CSM carried out on designated sites. CS data will be biased toward recording the more common and widespread CSM Indicator species in the same way that it's sampling design leads to poor representation of the rarest British habitats.

Status and recent change in 'appropriate diversity'

Widespread decreases in richness of both positive and negative indicators were seen between 1998 and 2007. A significant fraction of this temporal variation could be explained by nitrogen deposition, vegetation change along succession and productivity gradients and climatic trends although in many cases the directions of change seen were not consistent with expectation. The result is that most of the change in appropriate diversity could not be explained. This may not be surprising for a number of reasons. The differences analysed were based on two snapshot surveys and, whilst the best available explanatory variables were assembled, the analysis could not test for the effect of a number of potentially important additional driving forces. Ongoing loss of total taxon diversity in many Broad Habitats in Britain is a recurring theme across the CS time series albeit with the notable exception of arable land starting in 1990 reflecting the probable impact of setaside. Possible additional mechanisms driving this large-scale reduction in diversity could include the delayed ongoing effects of habitat fragmentation initiated much earlier in the 20th century leading to slow species loss but where species filtering is non-random (e.g. Lindborg and Erikkson 2004). Thus rare species and those vulnerable to eutrophication and changes in disturbance regime are more likley to decline than habitat generalists favoured by modern land-use regimes (Walker 2003; Smart *et al.* 2006b; Preston *et al.* 2002). Petit *et al.* (2004), for example, showed how patch size and shape, and interpatch distance could explain part of the spatial variation in Ancient Woodland Indicator species richness in woods sampled in CS indicating that such spatial signals are there to be found. While not feasible within this project, further analyses are possible extending hypotheses to test for patch geometry effects onto temporal change among a wider range of habitats.

Since reduced appropriate diversity of indicator species was also detected in habitats extensively developed in the uplands which are less likely to have been historically fragmented, including Bog, Dwarf Shrub Heath and Acid Grassland, it seems that further explanations for diversity loss should also be entertained. Within-survey effects cannot be ruled out. The 2007 Quality Assurance (QA) survey found that fewer species were on average recorded than in 1998 (Wallace and Prosser 2008) but closer inspection of the data indicated that the difference was not statistically significant (Scott *et al.* 2008). Impacts associated with differences in the weather in each year of survey also have the potential to influence species richness but again, analysis of the most recent survey alongside data from the Environmental Change Network ruled out significant effects (Scott *et al.* 2010).

Explaining change; issues and constraints

Other constraints on our ability to attribute signals of change in appropriate diversity include a lack of data on management impacts. While it is unlikely that analyses seeking to explain changes in CS data will ever be based on comprehensive plot level management histories, the uptake of management prescriptions under increasingly widespread agri-environment schemes is recorded in detail by the agencies involved. Despite attempts in this project to secure geographically comprehensive but fine-scale data on polygon management history and current prescriptions, only current polygon-level ELS data for England was available. The absence of data that tracks positive maintainance and restoration of habitats in the wider countryside means that further attribution analyses may miss signals of beneficial ecological changes and risks producing unbalanced messages that overly dwell on the impacts of negative drivers because datasets that track these drivers are more readily available.

An additional explanation for the lack of consistency in the results for CSM Indicator richness concerns properties of the response variable rather than aspects of the explanatory variables. The pools of species in each Priority Habitat list often consist of a mix of traits and ecological affinities so that different directions and sizes of response to the same stressor might have occurred. Even so, positive indicators are clearly distinguished on ecological grounds from the more nutrient-demanding generalists of the negative group (see Box 4.1) so that any explanation for the general reduction in plot richness needs to account for impacts that affect both groups.

Changes in negative indicator cover were more consistent with expectation. On average, reductions in cover were detected in Neutral Grassland, Dwarf Shrub Heath and Bog but negative indicator cover increased in Acid Grassland and the two linear Broad Habitats between 1998 and 2007. In both linear habitats, cover was on average much higher than in the other Broad Habitats. Correlated drivers of the change in negative indicator cover showed that anthropogenic stressors such as atmospheric deposition of reduced nitrogen and climate warming since 1980, had expected positive effects but increasing succession between 1990 and 1998 was associated with reduced cover between 1998 and 2007 in Calcareous Grassland, Neutral Grassland and Rivers and Streams. The apparently counterintuitive relationship between negative indicator cover and vegetation succession could be due to growth having proceeded to the point where even nutrient-demanding yet herbaceous negative indicators are debilitated by further shading from shrubs and trees. Exploring this hypothesis would require further analysis of the data.

Arguably, the most important conclusion from the attribution analysis is that factors outside the immediate control of site managers have shown demonstrable correlated impacts on the abundance of species (negative indicator cover) that reduce the conservation value of habitats. This highlights the need for better modelling of the expected effects of these drivers to aid conservation planning as well as providing a context for better evaluation of site-based monitoring (Rowe *et al.* 2009; van Dobben and Wamelink 2009 and see below).

4.9 Modelling applied to 'appropriate diversity': changes in habitat suitability for *Sphagnum* on upland peat soils in the United Kingdom in response to pollutant deposition and climate change

Introduction

This section summarises the development of two statistical models and their subsequent application in forecasting change in a specific CSM Indicator species group (ombrotrophic *Sphagnum* cover) in a particular Priority Habitat (Blanket Bog) in upland Britain in response to climate change and atmospheric nitrogen deposition. Full details of the work are given in Appendix 4.3. An uncertainty analysis is also presented comparing the performance of two modelling techniques. This is consistent with the current best-practice of ensemble forecasting where more than one modelling technique is used to generate an envelope of predictions. This recognises that often no one particular technique is recognisably the best (Araújo and New 2006). The approach illustrated is an example of the use of chained process models to produce outputs for soil and climate variables, which are then used as input into static, empirical niche models for individual species. The approach is part of a Europe-wide research effort motivated by the need to model the impacts of atmospheric pollutant deposition on soils and biodiversity (DeVries *et al.* 2010). In Britain, activity has centred on the production of an ensemble of niche models for higher and lower plants (Smart *et al.* 2010). Here we generated new niche models for ombrotrophic *Sphagnum* species based on recorded cover in Countryside Survey plots.

Ombrogenous mire ecosystems are classified as Priority Habitats (Blanket Bog and Raised Bog) under the United Kingdom Biodiversity Action Plan and as Annex II habitats under the EEC Habitats Directive. The first step is to model the realised niche of aggregated ombrotrophic *Sphagnum* species using paired species abundance and environmental data from Britain. Two popular techniques, Generalised Linear Mixed Models (GLMM) and Generalised Additive Mixed Models (GAMM), were used to generate two empirical niche models. Scenarios of climate change and pollutant deposition for the years 2020, 2030 and 2050 were then applied singly and in combination to drive changes in the predicted suitability of the niche for ombrotrophic *Sphagnum* species in upland mires of the UK. Our objective was to model impacts at the three decadal time steps across the UK, and then to evaluate the relative contribution of the uncertainty on the parameters of the empirical niche models by comparison with other sources of variability in the dataset of predictions. Specifically we asked the following questions:

1. What are the best predictors of ombrotrophic *Sphagnum* cover across Britain?
2. What are the relative contributions of the following to the variability in model predictions; a) modelling technique, b) niche model parameter uncertainty, c) variation in climate predictions, d) predicted change over time conditional on each scenario, e) spatial variation in predicted *Sphagnum* cover across the UK.
3. Does predicted change in *Sphagnum* cover due to climate vary across the UK and how big is the predicted temporal change compared to spatial variation in habitat suitability?
4. Is climate or pollutant deposition a more important driver of predicted change in habitat suitability?

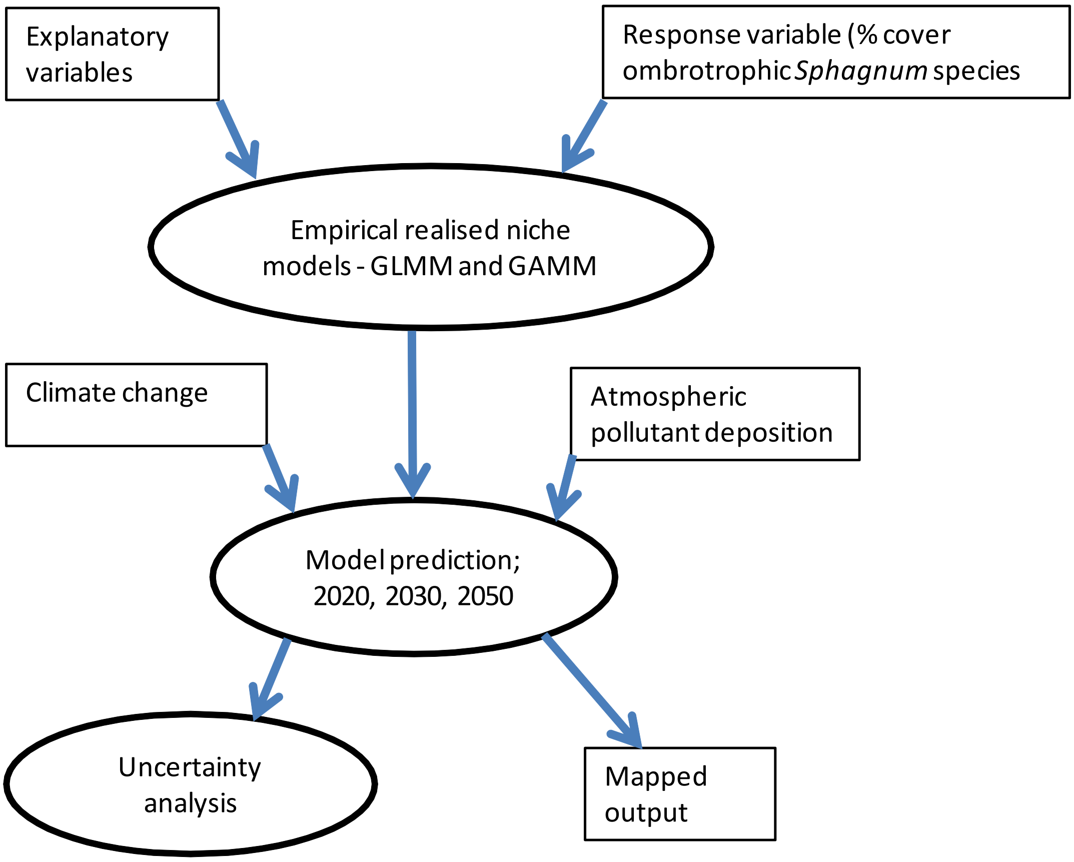
Again it should be emphasised that this is just one example of analyses that can be applied to many other higher and lower plants (Smart et al. in press). Since 74% of CSM species have existing niche models, there is ample scope for extending scenario testing to a significant proportion of those species most important in evaluating appropriate diversity in British ecosystems. We focus on *Sphagnum* cover in Blanket Bog since CSM guidance for this Priority Habitat indicates that *Sphagnum* cover should be present comprising species other than the relatively pollution tolerant *S. recurvum*.

Methods

Analysis was carried out in two stages (Fig. 4.5); niche models were first produced for *Sphagnum* species. These models were then used to forecast species cover in 2020, 2030 and 2050 by driving them with inputs from climate change scenarios and inputs of soil variables derived from simulating the impact of sulphur and nitrogen deposition on the soil biogeochemistry of upland peats. This simulation was carried out by using modelled estimates of atmospheric pollutant deposition from the Fine Resolution Atmospheric Multi-pollutant Exchange Model (FRAME) (Fournier *et al.* 2003) as input to the soil biogeochemical Very Simple Dynamic (VSD) model (Posch and Reinds 2008). Modelling of deposition was carried out at the 5km^2 scale and soil responses modelled at the 1km^2 scale. *Sphagnum* cover was modelled at the scale of 4m^2 plots but averaged over 1km^2.

Climate change scenarios were based on UKCP09 and UKCIP02 datasets. Given concerns over the spatial coherence of the UKCP09 gridded forecasts, the UKCIP02 scenarios were run in parallel and compared the results to validate. Our analysis and conclusions focused on the UKCP09 datasets because these included uncertainty estimates on the ensemble distributions for each grid square. For UKCP09 variation in the average predictions for each time interval and across the 25km^2 is available from the website and predictions were based on climate values at the 33%tile, 50%tile (the central estimate) and the 67%tile. The full uncertainty within each RCM ensemble member is not included in these outputs while the effect of averaging at the 25km^2 scale will also have the effect of reducing the total range of the predicted values and rendering the predictions insensitive to probable variation in climate change below the 25km^2 grid (Trivedi *et al*. 2008). UKCIP02 projections were also acquired from *ukcip.org.uk* for comparison with UKCP09. Whilst these forecasts are on a 5km^2 grid no uncertainty estimates are attached to them. Data was downloaded for the mean daily maximum July temperature, the mean daily minimum January temperature and mean annual rainfall.

Figure 4.5: Flow diagram of analysis. Rectangles indicate input or output data. Ellipses indicate analytical processes.



Results

What were the best predictors of ombrotrophic Sphagnum cover across Britain?

The best fitting GLMM and GAMM models differed in the explanatory variables selected. The GLMM model included a quadratic relationship with vegetation canopy height indicating an optimum for ombrotrophic cover in short vegetation but less likely to be found in the shortest vegetation and not favoured under taller canopies. Other terms reflected expected positive responses to substrate C:N ratio and mean monthly rainfall per year based on a significant interaction between the two. This indicated that increase in the favourability of conditions for *Sphagnum* at higher C:N and rainfall was greater than just the sum of the two variables.

Trying all possible combinations of parameters together with hypothesised interactions, resulted in a best-fitting GAMM model that consisted of terms for percent carbon in the soil, vegetation height, two dimensional terms for the interaction of annual rainfall and percent carbon, and the interaction of mean monthly rainfall per year and the mean maximum temperature in July, an interaction of the spatial location of each 1 km square and a random effect of survey square on between-plot variation.

In summary, both models expressed the preference of ombrotrophic *Sphagnum* for short vegetation, with a very wet, cool climate associated with low decomposition rates and hence high carbon accumulation.

Explanatory power of the minimum adequate models

As expected the GAMM model explained substantially more variation in the training data (Table 4.5). In particular, the ability of the GAMM smoothing functions to account for local variation in the data space was evidenced by the much higher amount of between-plot, within-square variation accounted for. The proportions of between-square variation explained by each model were more similar. Despite the better performance of the GAMM, it still only managed to explain 51% of the variation in the cover data used to build the model (Table 4.5).

Table 4.5: Decomposition of total variance in ombrotrophic* Sphagnum *cover explained by each modelling technique based on the same training data from the Countryside Survey carried out in 1998 (n=623 vegetation plots in 172 1 km squares).

% Variance explained	Between 1 km sqr	Within 1 km sqr	Total
GAMM	77.1	48.1	51.2
GLMM	69.1	16.9	37.1

4.10 Discussion

Does predicted change in Sphagnum cover due to climate and pollution vary across the UK?

Both models predicted either stability or a decrease in ombrotrophic *Sphagnum* cover across the UK between 2020 and 2050 (Figs 4.6 and 4.7). However, all predicted changes were generally small and uncertain; much more so for the GLMM. The GLMM predictions did however, take into account pollutant deposition in addition to climate change and so it is useful to compare the two model outputs on the basis that where they agree spatially, the much less uncertain GAMM predictions should provide a cross-validated impression of where climate-induced change maybe more important than pollutant-induced change.

Peatlands predicted to be most negatively impacted by climate change in both models were found in northern Scotland scattered through Assynt, Wester Ross and down to Lochaber. Bogs on Mull and neighbouring Morvern were also expected to be impacted in both niche models as were areas of bog in the south of Scotland, in Galloway and to the south of Peebles. In England and Wales both models predicted that the largest reductions in ombrotrophic *Sphagnum* cover would be in the western Lake District and the Brecon Beacon area of south Wales (Figs 4.6 and 4.7). Areas that were only predicted to be impacted by climate change by 2050 in the GAMM model were Dartmoor, peatlands on the western border of Northern Ireland and north Lewis in the Outer Hebrides. Comparison with the GLMM predictions of changing *Sphagnum* cover in response to climate and pollution highlight areas where pollution seems to be the more important driver – Forest of Bowland and peatlands in Wales – or where pollution exacerbates a predicted impact due to climate change – Dartmoor, Brecon Beacons and the western Lake District.

Predictions of *Sphagnum* cover in 2050 were made using both UKCP09 and UKCIP02 datasets (Figs 4.8 and 4.9). Although spatial patterns in the predictions were very similar, the most obvious difference relates to the averaging of predicted values within the large 25km grid squares used by UKCP09 as opposed to the 5km UKCIP02 grid. This inevitably reflects the averaging of the climate predictions in the larger UKCP09 grid squares. This effect will be influential in squares that vary greatly in the presence of upland terrain since climate predictions may not reflect this local variability. The advantage of the UKCP09 dataset is that the GB-wide datasets do allow exploration of the uncertainty around the distributions of projections albeit that these projections lack precision where variation in topography is marked within the 25km squares.

Prospects for better modelling of Sphagnum and other CSM Indicators

For an ecosystem dominant such as *Sphagnum,* it may be feasible to build a growth model that dynamically represents its response to favourable conditions including a hydrological component and a plant competition component. However, if the aim is to model the suitability of conditions for a much larger range of species of importance to conservation then it is not feasible to parameterise a model per species (cf. Terry *et al.* 2004). The middle ground is to develop a growth model based on a realistic yet small number of Plant Functional Types (PFT). Then the outcomes of modelled competition between the PFT can be coupled with abiotic and climate data to solve empirical niche models (DeVries *et al.* in press; Wamelink *et al.* 2005). These still have shortcomings but can still act as useful scenario testing tools (e.g. Wamelink *et al.* 2003). This depends upon quantification and effective communication of uncertainties and a clear understanding of the implications and shortcomings of using a spatial niche model as a source of hypotheses about temporal change, including the risks of extending a static niche model into novel configurations of environmental space not sampled in the training data (eg. Broenimann *et al.* 2007).

Niche modelling – the benefits of using more than one modelling technique

In the last ten years, much research effort has been focused on modelling the niche of species of plants and animals. This focus has been stimulated by a resurgence of interest in the niche concept and the urgent need for informative forecasts of the response of taxa to climate change alongside other global change phenomena (Bakkenes *et al.* 2002; Harrison *et al.* 2006). A wide range of techniques are now in use (Ellith *et al.* 2006) yet given that no single front runner has emerged, good practice is to apply a number of modelling methods including their attendant uncertainties to produce ensemble forecasts. The idea is that a consensus prediction from a number of separate models can more reliably delimit uncertainty bounds and likely trajectories than a single model. We applied two robust and routinely applied techniques. They offered useful and complementary perspectives on the uncertainties involved in modelling *Sphagnum* cover as well as the possible locations of maximum vulnerability to climate change and pollutant deposition up to 2050. The GAMM as expected outperformed the GLMM in terms of explanatory power (Table 4.5). This is because the locally weighted smoothing functions are very effective in capturing local differences in the form of the relationship between response and explanatory variables as well as residual spatial correlation linked to the proximity of the observations but not explained by the explanatory variables. In the GLMM local variability is relegated to residual error since the systematic part of the model can only fit a global linear or quadratic response. The tendency of GAMM to overfit training datasets can sometimes result in a typically better fit to observed data than GLMM being accompanied by poorer transferability than GLMM when applied to the same species in a different region (Randin *et al.* 2006). In the current application a probably more robust approach would be to model presence rather than cover. This would avoid overfitting to variation inadequately tracked by available explanatory variables because at least part of the variation is most likely attributable to sampling error and weather impacts at the time of survey in addition to ecologically more meaningful factors in terms of the *Sphagnum* niche.

The need for more integrated scenarios of change in multiple drivers

Human activities that produce Greenhouse Gas (GHG) emissions also produce eutrophying and acidifying nitrogen and sulphur. Realistic modelling of the impacts of multiple drivers ought to start with integrated scenarios of change in driving variables that jointly project emissions of chemically active and radiatively active gases plus associated land-use and land-cover change. It seems that exactly this kind of work is underway in the production of new Representative Concentration Pathways that unify the contributions and needs of climate modelling, integrated assessment modelling and analysis of impacts, vulnerability and adaptation (Moss *et al.* 2008). However, these are not due for publication until 2013.

Figure 4.6: Predicted change in % *Sphagnum* cover per 2x2m plot (2020 to ’50) using the GAMM. Predictions driven by changes in climate data only.

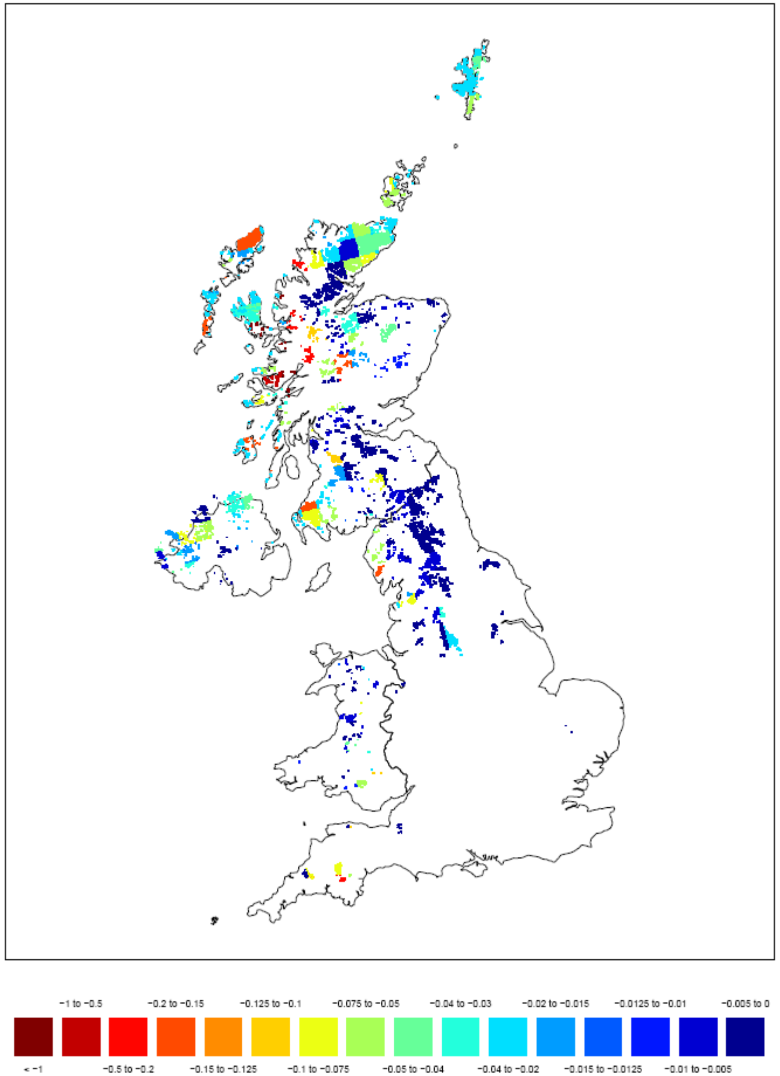


Figure 4.7: Predicted change in % *Sphagnum* cover per 2x2m plot (2020 to ’50) using the GLMM. Predictions driven by changes in climate data and pollution data.

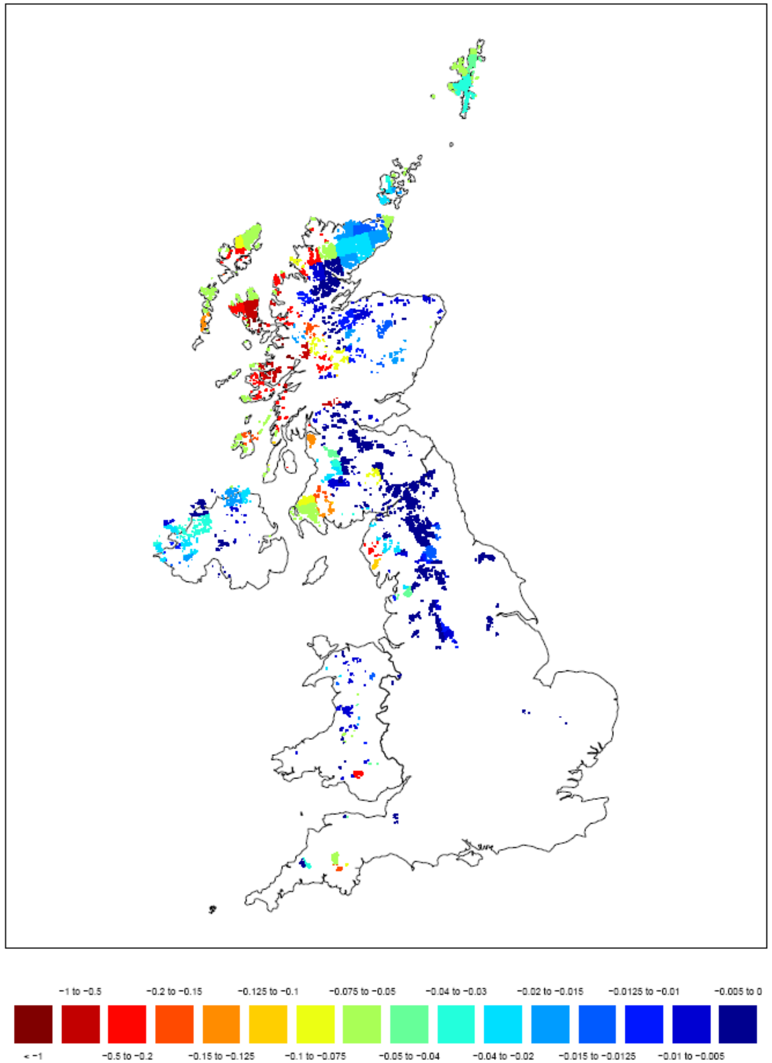


Figure 4.8: GAMM predictions of potential* Sphagnum *cover per 2x2m plot for 2050 based on median estimates of UKCP09 climate variables. Maps show central (b), upper (c) and lower (a) estimates based on the niche model parameter estimates and their 95% Confidence Intervals.

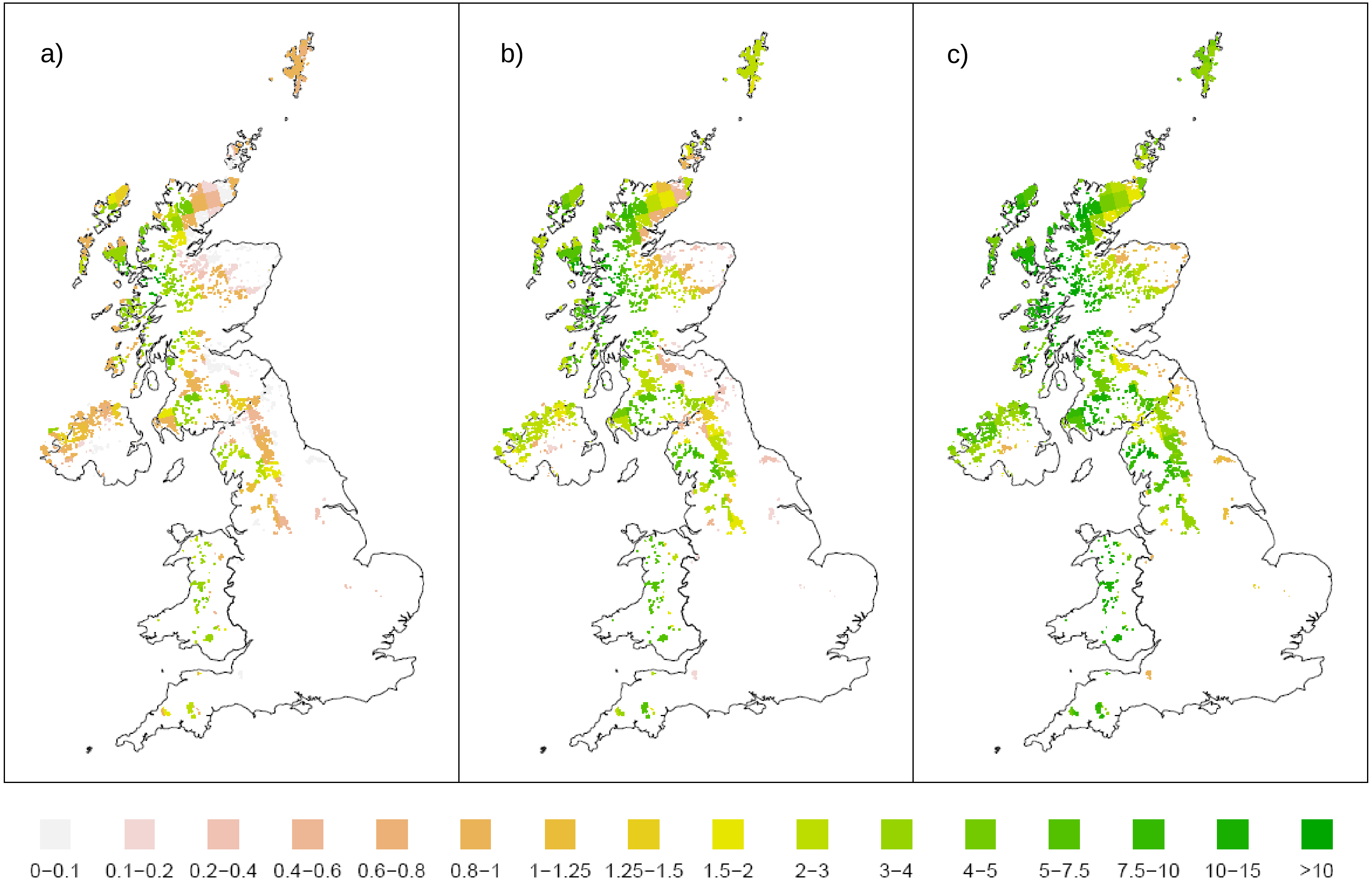
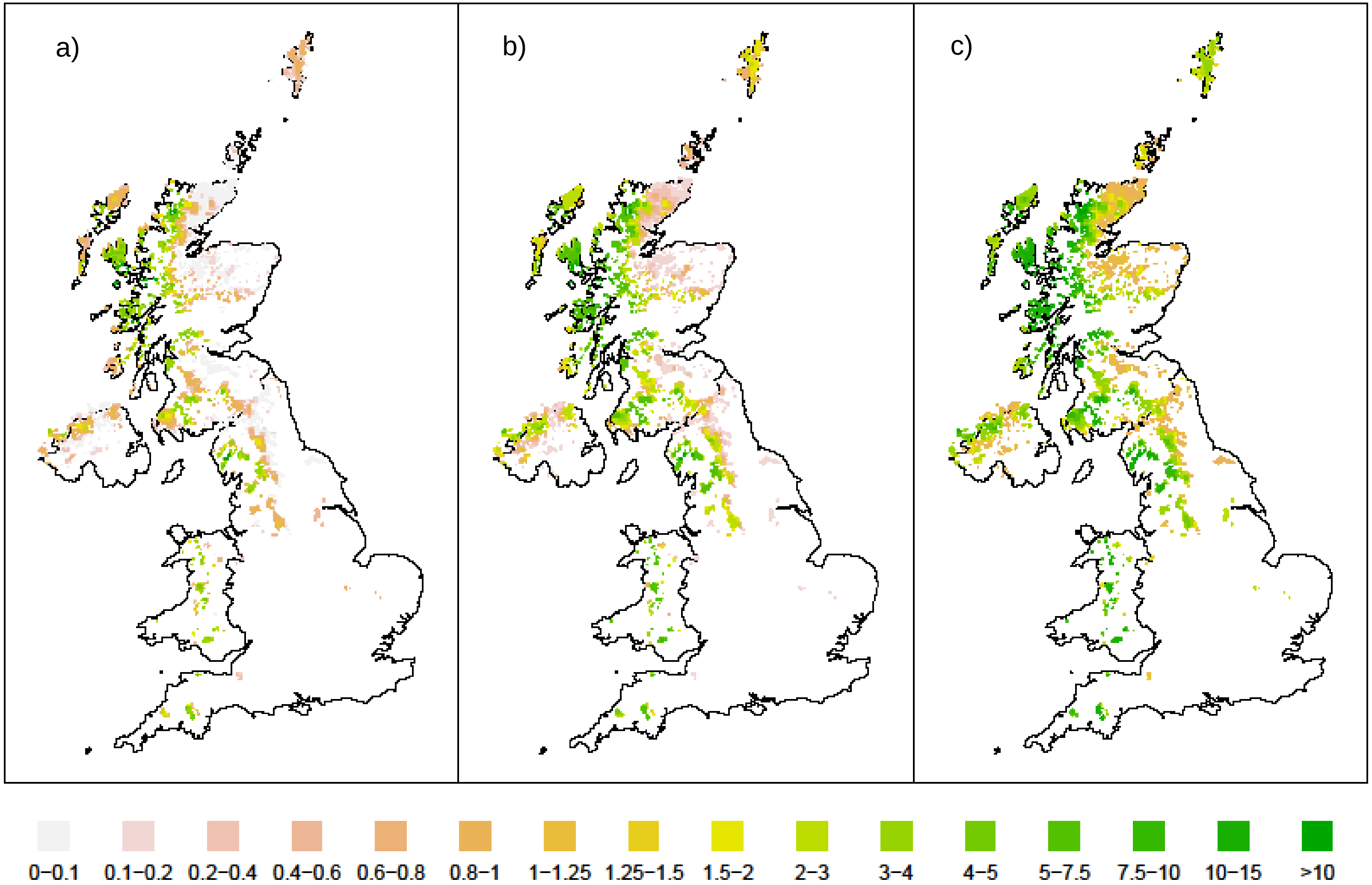


Figure 4.9: GAMM predictions of potential* Sphagnum *cover per 2x2m plot for 2050 based on UKCIP02 climate variables. Maps show central (b), upper (c) and lower (a) estimates based on the niche model parameter estimates and their 95% Confidence Intervals.



Chapter 4: Appendices

4.1: Tabulated frequancy of Common Standards Monitoring Indicator species in CS plots.

4.2: ITE Land Class maps showing the mean richness of positive and negative CSM Indicators per areal (X Plot nest 2 and U Plots both $4m^2$) subtracted from the mean richness per Linear Plot ($10m^2$) or Targeted Plot (Y Plots – $4m^2$). Also shown are maps of the standard error of the difference across each land class. Available online.

4.3: Report on development and application of a small ensemble of two models for a CSM Indicator species group; ombrotrophic *Sphagnum* spp. Draft paper submitted to EA-QUEST special issue of Climate Research and in revision. Contact Simon Smart (ssma@ceh.ac.uk) for the latest version.

Chapter 5: Nectar plant diversity; an indicator of the 'regulating' ecosystem service of pollination

S.M. Smart, P. Henrys

Summary

- Reductions in nectar plant diversity were widespread in Britain between 1990 and 2007 but they were mainly confined to small fragments of semi-natural habitat (sampled by the Targeted Plots) embedded in larger areas of common Broad Habitats (sampled by Main Plots and Unenclosed Plots[29]) . Larger magnitude changes and hence bigger potential impacts on the pollination ecosystem service are likely to have happened before 1990. Estimating the impact of recent and historical declines on crop pollination requires further calibration of observed changes against study systems in which coupled changes in plants, pollinating insects and fruit set have been observed.

- Counts of nectar-providing plant species in Countryside Survey (CS) vegetation plots were used as an indicator of potential pollination service delivery in British habitats. Analyses focused on nectar plants for bees (bumblebees and solitary bees combined).

- The highest mean numbers of nectar plants per plot in Broad Habitats in 2007 were found in; Calcareous Grassland (12 per 200m^2 Main Plot), Rivers and Streams (5 per 10 m^2 Linear Plot), Boundary and Linear Features (6 per 10 m^2 Linear Plot), Neutral Grassland (4 per 200 m^2 Main Plot) and Broadleaved Woodland (3 per 200 m^2 Main Plot), (note that CS does not provide representative coverage of coastal nor urban habitats).

- Most changes (declines between 1990 and 2007) occurred in small semi-natural habitat patches embedded in common Broad Habitats and on stream and ditch banks. Declines were largest in Arable and Horticulture, Neutral Grassland, Broadleaved, Mixed and Yew woodland and Coniferous Woodland, which all lost on average 1 species per 4m^2 Targeted Plot in the 17 year period. Few changes were detected in larger areas of common Broad Habitat between 1990 and 2007.

[29] For definition and nomemclature of vegetation plot types sampled in CS, see CAREY, P.D. et al. (2008) Countryside Survey: UK Results from 2007. Online at: www.countrysidesurvey.org.uk.

- A range of explanatory variables were applied to try and explain spatial differences in change in nectar plant diversity over time. No signals of climate change or pollutant deposition were detected.

- Succession was important in suppressing nectar plant diversity between 1990 and 2007 on Rivers and Streams and in woodlands. Being a feature of late-successional habitats increasing vegetation growth was likely to be associated with woodland gap closure and increases in woody cover on streamsides. Boundary and Linear Features showed the reverse effect; successional change in the 17 year period favoured more nectar plants per plot.

- Change in species composition along an inferred vegetation productivity gradient was associated with either increases or decreases in nectar plant diversity. When vegetation moved toward less productive species, infertile habitats tended to lose diversity while productive habitats gained nectar plant diversity.

- A strong negative correlation between nectar plant richness and sheep density was seen in upland Dwarf Shrub Heath and to a lesser extent in Bog and Fen, Marsh and Swamp.

- Statistical models of spatial variation in nectar plant diversity for bees and butterflies across British Broad Habitats were constructed. The best fitting models included Broad Habitat, % woody cover, climate variables, nitrogen deposition, length of linear features and other landscape attributes as explanatory variables. These models were used to produce predictive maps of nectar plant diversity across Britain and to test the impact on nectar plant diversity of the Defra “environment-only” scenario of agri-environment scheme impacts in English Severely Disadvantaged Areas.

- The modelling work was highly novel and showed much potential for exploring multiple impacts of human activities at the fine scale across British habitats. Similar models have been constructed for Common Standards Monitoring Indicator species and an indicator of above-ground Net Primary Production.

5.1 Introduction

Globally, it has been estimated that 35% of food production from crops depends upon animal pollination (Ricketts *et al.* 2008) while Williams (1994) estimated that 84% of European crops depended at least partly on insect pollination. Domesticated honey bees provide a critical source of managed pollination but their numbers have been threatened by disease and pesticides (Ricketts *et al.* 2008). Wild pollinating insects can provide significant pollination services (Kremen *et al.* 2002; Allen-Wardell *et al.* 1998) but wild pollinators have also declined, most likely in response to agricultural intensification, which results in simplified landscapes lacking the diversity of habitats and nectar plants associated with high pollinating insect diversity (Heard *et al.* 2007; Carré *et al.* 2009; Ricketts *et al.* 2008; Carvell *et al.* 2006a).

In this chapter, the diversity of nectar plants in the vegetation plots of the Countryside Survey (CS) is quantified at the national scale by Broad Habitat. Current status and past changes in mean nectar plant diversity are quantified and presented as a context for evaluating changes in the potential delivery of the pollination service across Britain. We then apply a range of explanatory variables to try and explain changes in nectar plant diversity across the countryside between 1990, 1998 and 2007. In the final section a spatial regression model is developed for predicting nectar plant diversity at the habitat patch scale in terms of Broad Habitat, climate, landscape attributes and management status. Finally, the capability of the model for testing scenarios of land-use change is demonstrated by application to the Defra “environment only” scenario applicable in Severely Disadvantaged Areas in Britain.

5.2 Biophysical measurement

Higher plant species have been counted in fixed vegetation plots in successive Countryside Surveys. By matching these records with a list of known nectar-providing plants, change in the numbers of plant species providing nectar resources for pollinating insects can be quantified between surveys and across Britain by Broad Habitat within each survey. Broad Habitat mapping, in conjunction with plant species data, is at present available for the 1990, 1998 and 2007 surveys and so patterns of spatial and temporal change in nectar plant diversity are presented for these surveys. In this chapter we mainly focus on nectar plants for bees however both butterfly and bee nectar plant diversity are considered in the modelling section. The emphasis on bees reflects the fact that, aside from managed honeybee colonies, wild bee populations are the most important pollinators of crop monocultures (Klein *et al.* 2007; Ricketts *et al.* 2008)

Defining nectar plant diversity

Lists of nectar plant species for bumblebees and solitary bees were provided by colleagues at CEH in Wallingford[30] and included sources listed in Carvell *et al.* (2006a). The numbers of these plant species in each plot were summed to give a count per vegetation plot. Subsets of count data were prepared for analysis. Depending on the analysis question, these subsets were grouped by Broad Habitat, plot type and landscape location.

[30] Lists of nectar-providing plants were based on floral visitation survey data for Britiain compiled from a wide range of sources by Prof Richard Pywell and Dr Claire Carvell.

The resulting counts of bee nectar plants were assembled for each Broad Habitat with the following modifications applying:

1. Counts were not weighted by visitation data. These data have been previously assembled for some species (see Carvell *et al.* 2006a) and are being finalised for the complete list used here. Weightings can be applied in the future to evaluate differences in species composition given differences in richness.
2. Diversity indices were not analysed – see section 5.3.
3. Coastal and urban habitats were excluded since CS does not provide representative coverage of these Broad Habitats. These are significant omissions since these habitats are known to be particularly important sources of nectar plants and bee diversity in Britain.
4. Montane and Inland Rock were not analysed because of their limited value as bee habitats and suppliers of nectar resources.

5.3 Rationale for selection

Countryside Survey does not measure the abundance of bees or butterflies but does measure higher plant species composition in an unbiased sample of fixed plots across common British habitat types. The response variable derived from these data is simply the numbers of nectar providing plant species per plot. Our analysis focuses on species richness rather than using diversity indices because this is consistent with measuring change in the species pool as a subset of total biodiversity. This amounts to measuring the functionally important fraction of total natural capital that can contribute to pollination services for crops and wild plants. In any particular place it may be specific plants that are important for particular insects but even a cover-weighted index maybe a poor correlate of actual resource availability if management prevents flowering. For example in Improved Grassland in 2007, the most frequent nectar providing plants were *Trifolium repens* and *Ranunculus repens* (Appendix 5.1). In these frequently mown grasslands, cover estimates for these clonal species are likely to be only weakly correlated with floral abundance. A detailed habitat-specific and food-web centred perspective is outwith the scope of this analysis. Instead, analysis of large-scale but fine grained pollinator plant diversity provides a useful context for more detailed studies. For example, Beismeijer *et al.* (2006) analysed changes in 10km square richness of insects and pollinator plants across Britain and the Netherlands and suggested a causal link between declines in the two groups. Similarly, in this chapter we first quantify change in the numbers of nectar providing plants per CS plot and then go on to try and explain temporal changes and model spatial patterns in terms of potential causal factors.

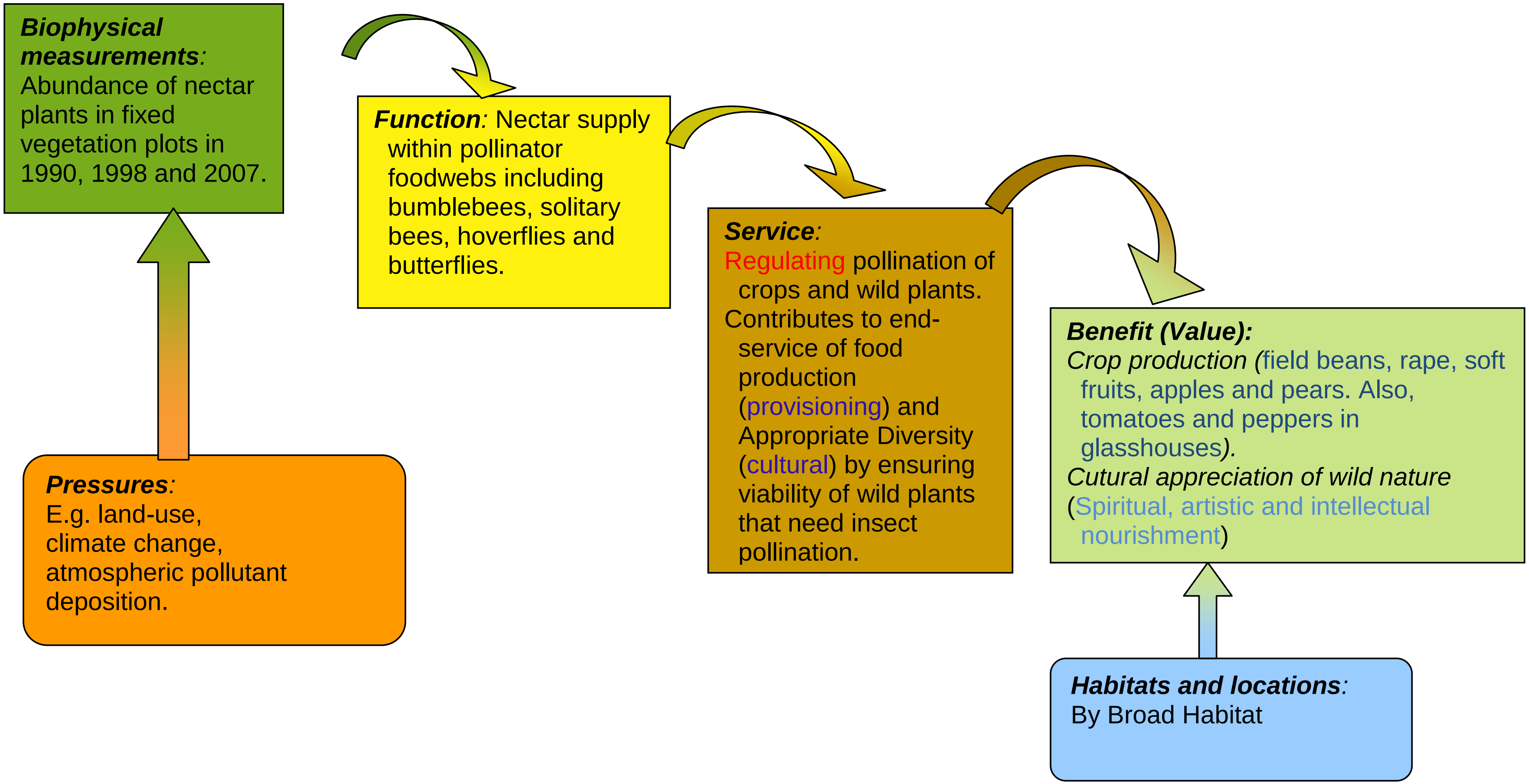
5.4 How is nectar plant diversity linked to ecosystem services?

Nectar resources provide the carbohydrate-rich reward for insects that transfer pollen between individual plants ensuring sexual reproduction and the production of viable fruit and seed. A range of crop species and wild plants depend upon obligate out-crossing via insect-mediated pollen transfer (Klein *et al.* 2007)[31]. Without compensatory increases in availability of domesticated bees, loss of wild pollinators, as a result of landscape simplification and intensification, has been linked to reductions in crop pollination services (Kremen *et al.* 2002; Klein *et al.* 2007). Conversely, maintenance of pollinator diversity, favoured by higher habitat and nectar plant diversity, has been shown to stabilise levels of crop pollination (Klein *et al.* 2007; Ricketts *et al.* 2008). Since nectar resources are essential for the maintenance of wild bee populations decline in their diversity amounts to a reduction in the natural capital required to support pollinator foodwebs and the delivery of the pollination service. Declines in nectar plant abundance are therefore a relevant indicator of reduced pollination service provision. This interpretation applies irrespective of whether declines in bees drive loss of obligate out-crossing plants or whether loss of nectar plants drive reductions in wild bee population sizes (Beismeijer *et al.* 2006).

Analysis of spatial and temporal patterns of change in diversity of nectar plants therefore provides contextual information on the integrity of one trophic level essential to the functioning of pollinator food webs and to the delivery of the regulating ecosystem service of pollination (Carvell *et al.* 2006a). This service in turn provides support for the provisioning of food production and, by helping maintain the viability of pollinator foodwebs among wild plants and bees, supports the cultural service of wild species diversity (Fig. 5.1).

[31] Obligate out-crossing plants are those where viable fruit or seed is only produced if pollen is exchanged between different plants. While many plants are wind-pollinated, others require insects to transfer the pollen.

Figure 5.1: The ecosystem service cascade for Nectar Plant Diversity (after Haines-Young and Potschin 2007).



5.5 Current status and trends across GB

Change in nectar plant diversity between 1990, 1998 and 2007

Analysis of change was carried out by Broad Habitat using the consistent modelling approach applied in previous reporting for Countryside Survey 2007 (Scott, 2007). This ensures that all the data available in each year were used, thereby increasing the power of the estimation procedure by using both repeated and non-repeated plots. The model simultaneously generates estimates of mean richness within each year and tests the significance of the differences between the three surveys (Figs 5.2 and 5.3).

The only significant changes in nectar plant diversity to have occurred in larger areas of common habitat and unenclosed land were in Arable and Horticultural (increase between 1990 and 1998), Bog (decrease between 1990 and 2007) and Fen, Marsh and Swamp (decrease between 1998 and 2007) (Table 5.1, Fig. 5.2). Although statistically significant, all changes were relatively small in that their magnitude was never large enough to move the mean value into the range of a much more or less diverse Broad Habitat. For example, despite a significant reduction between 1998 and 2007, the mean nectar plant richness for Fen, Marsh and Swamp in 2007 was still much closer to mean Fen, Marsh and Swamp richness in 1990 than to the appreciably lower values typical of Bog, Dwarf Shrub Heath and Acid Grassland (Fig. 5.2). Similarly the increase in Arable and Horticulture between 1990 and 1998 did not move the mean richness anywhere near to typical values of Improved Grassland, Neutral Grassland or Boundary and Linear Features (Fig. 5.2).

Changes were also analysed across the population of small patches of semi-natural habitats located within larger areas of each Broad Habitat. These patches are diverse in their species composition and pick out atypically rich or distinctive assemblages not common enough in each 1km square to have been sampled by the Main Plots analysed above (Smart *et al.* 2006a). Examples include small flushes, bog pools, hydroseres around waterbodies, species rich grassland fragments in field corners or on steep slopes, coastal assemblages, species-rich weed communities, distinctive woodland gaps and rides. A common theme among such patches is their early to mid-successional status. This means that without appropriate disturbance species composition may well undergo directional change (Smart *et al.* 2003). In Countryside Survey these habitat patches are represented by the Targeted Y Plots (Carey *et al.* 2008).

Analyses of change in nectar plant richness in the Targeted Plots showed a more dynamic situation than in the Main Plots (Table 5.1). Declining nectar plant diversity was seen between at least one pair of surveys in small habitat patches (Targeted Plots) sampled within all Broad Habitats analysed except Acid Grassland, Calcareous Grassland – where sample size was small – and Fen, Marsh and Swamp. The changes of largest magnitude occurred in the three lowland agricultural Broad Habitats; Improved Grassland, Neutral Grassland and Arable and Horticulture, and in the two woodland Broad

Habitats (Table 5.1, Fig. 5.3). Observed changes in these small habitat patches were also substantially larger than in the Main Plots representing larger areas of common habitat and unenclosed land.

Table 5.1: Changes in mean nectar plant diversity by Broad Habitat between 1990, 1989 and 2007.

Broad Habitat	Landscape location[32]	1990 to 2007	1990 to 1998	1998 to 2007
Broadleaved, Mixed and Yew Woodland	larger areas of habitat	ns	ns	Ns
Coniferous Woodland	"	ns	ns	Ns
Arable and Horticulture	"	ns	** (↑)	Ns
Improved Grassland	"	ns	ns	Ns
Neutral Grassland	"	ns	ns	Ns
Calcareous Grassland	"	ns	ns	Ns
Acid Grassland	"	ns	ns	Ns
Bracken	"	ns	ns	Ns
Dwarf Shrub Heath	"	ns	ns	Ns
Fen, Marsh and Swamp	"	ns	ns	* (↓)
Bog	"	** (↓)	ns	Ns
Boundary and Linear Features	linear features	*** (↑)	* (↑)	** (↑)
Rivers and Streams	"	*** (↓)	*** (↓)	** (↓)
Broadleaved, Mixed and Yew Woodland	small habitat patches	*** (↓)	*** (↓)	** (↓)
Coniferous Woodland	"	*** (↓)	* (↓)	Ns
Arable and Horticulture	"	*** (↓)	* (↓)	Ns
Improved Grassland	"	*** (↓)	ns	** (↓)
Neutral Grassland	"	*** (↓)	ns	** (↓)
Calcareous Grassland	"	ns	ns	ns
Acid Grassland	"	ns	ns	ns
Bracken	"	* (↓)	* (↓)	ns
Dwarf Shrub Heath	"	ns	* (↓)	ns
Fen, Marsh and Swamp	"	ns	ns	ns
Bog	"	* (↓)	ns	ns

[32] Larger areas of habitat are sampled by the 200m^2 Main Plots and 4m^2 Unenclosed Plots. Linear features are sampled by Hedgerow, Field Boundary, Roadverge and Streamside Plots all 1x10m in size. This means that species richness values cannot be directly compared between plot types. See Smart *et al.* (2006a) for an approach to this problem and Carey *et al.* (2008) for further information on definitions of plot types.

Figure 5.2: Mean richness of nectar plants for bumblebees and solitary bees in 1990, 1998 and 2007 across Great Britain based on fixed vegetation plots from Countryside Survey in larger areas of habitat and unenclosed land (Main and Unenclosed Plots) and on linear Broad Habitats (Hedgerow, Roadverge, Field Boundary and Streamside Plots). BW=Broadleaved, Mixed and Yew Woodland, CW=Coniferous Woodland, B&L=Boundary and Linear Features, A&H=Arable and Horticultural, IG=Improved Grassland, NG=Neutral Grassland, CG=Calcareous Grassland, AG=Acid Grassland, Br=Bracken, DSH=Dwarf Shrub Heath, FMS=Fen, Marsh and Swamp, R&S=Rivers and Streams.

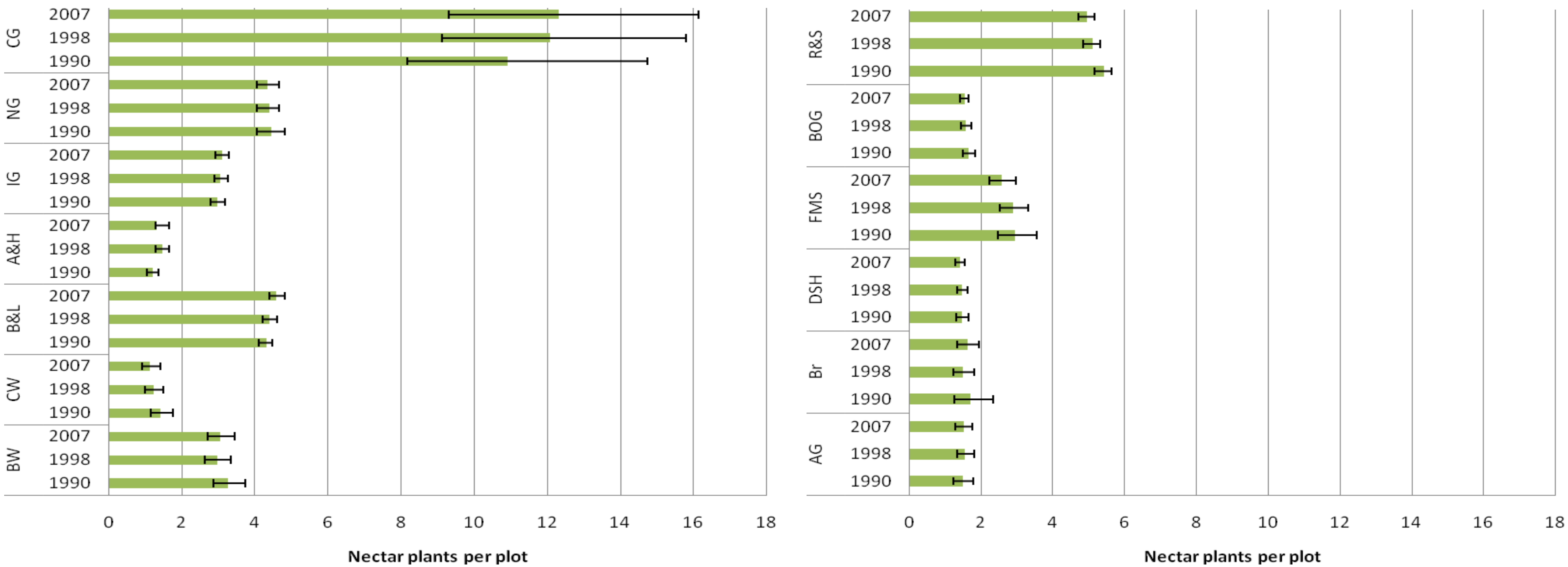
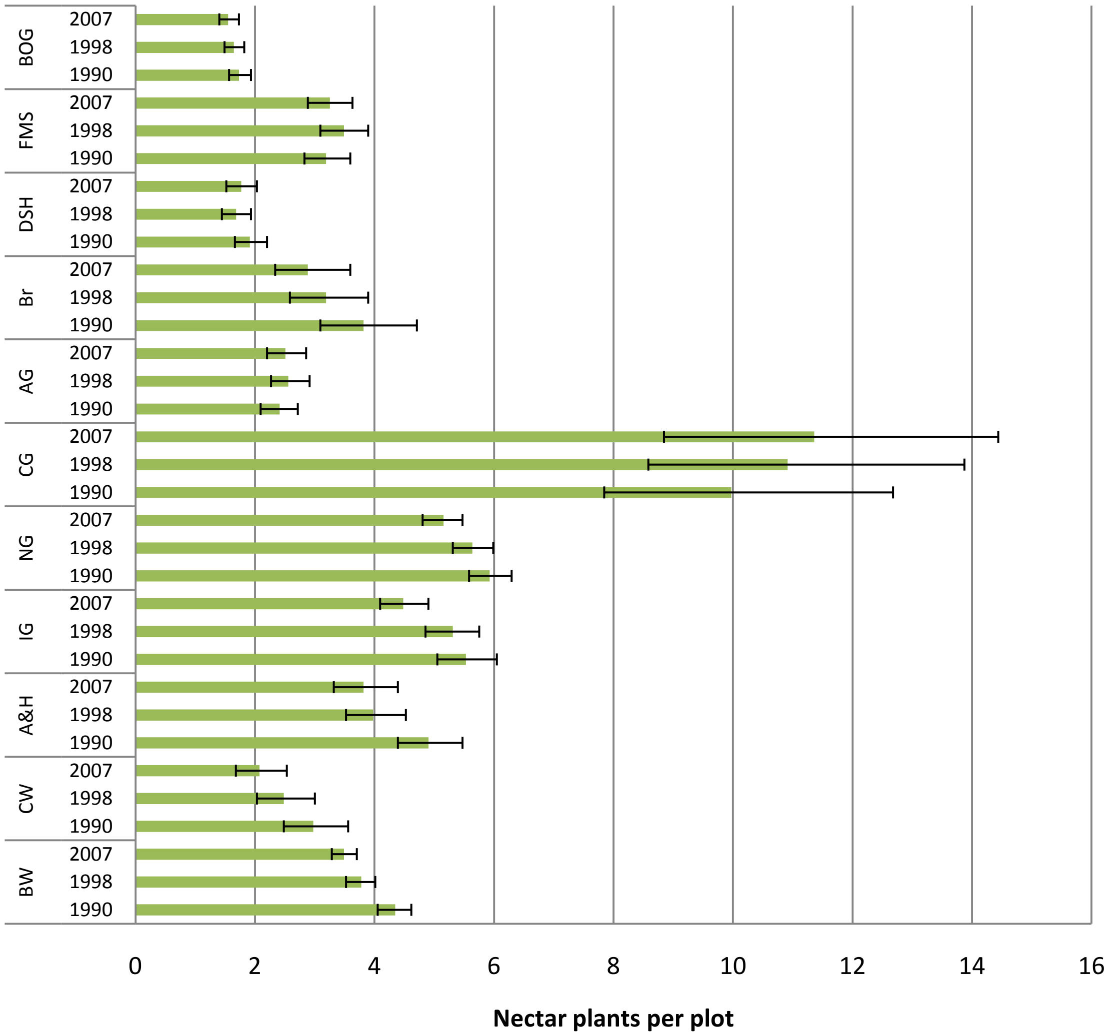


Figure 5.3: Mean number of nectar plants for bumblebees and solitary bees in Targeted (4m^2) Plots in 1990, 1998 and 2007 across Great Britain. Based on fixed vegetation plots from Countryside Survey targeted on small atypical habitat patches within larger areas of Broad Habitat.



5.6 Maps of current status

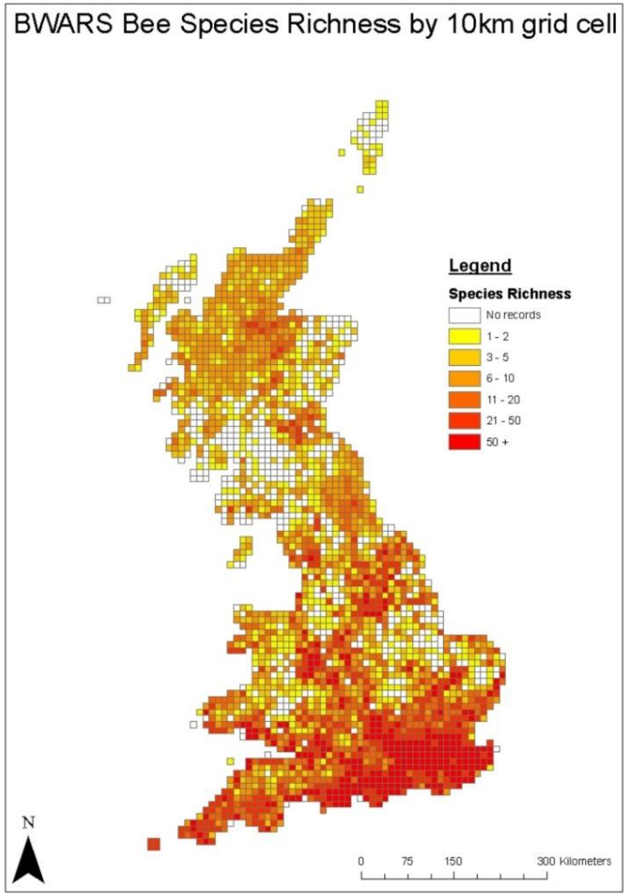
Pollinator insect and pollinator plant diversity across Britain reflect a range of controlling factors. Within grid squares, high diversity reflects the co-occurrence of diverse habitat types whilst low diversity reflects dominance by species poor habitats less favourable for bees and butterflies or their larval food plants and nectar resources (Heard *et al.* 2009). These habitat-related patterns in turn reflect climatic and soil-type constraints, for example Calcareous Grassland and dunes in the warmer south of Britain are associated with especially high pollinator plant and insect diversity (Williams 1985), while climate also constrains the distributional range of many pollinator insects within Britain.

When expressed as richness within 10km grid squares across Britain[33], a clear north-south gradient is apparent for bees, butterflies and hoverflies with the richest areas in the warmer south (Figs 5.4 and 5.5). By building a statistical model of nectar producing plant richness based on Broad Habitat and climate variables, predictions of nectar plant diversity can be produced for all British 1km squares (Fig 5.6). This provides an initial guide to the patterns of species richness expected at the national scale. It should not be treated as definitive as a full uncertainty analysis has not yet been conducted. Based on this statistical model the richness of nectar producing plants follows somewhat different patterns to those of bees, butterflies and hoverflies (Fig. 5.6). This partly reflects the scale of the mapping which is at 1km square and averaged across the Broad Habitats in each square. Moreover, mean nectar plant diversity per 1km square is also underestimated because linear Broad Habitats are not mapped by Land Cover Map 2000 (LCM2000) while patches of favourable habitat below the resolution of LCM (25 x 25m) will also be excluded. These aspects reduce the potentially important influence of small areas of very favourable habitat and linear features in increasing the total richness of nectar plants and pollinating insects in each grid square. Hence much of the east of England has low average nectar producing plant richness yet pollinating insect diversity is high (Fig. 5.6) reflecting the occurrence of favourable habitat within the dominant agricultural matrix (Carré *et al.* 2009). Nectar producing plant diversity is higher in the south, south west and in lowland Wales and through western England (Fig. 5.6). Scotland differs in mean nectar producing plant diversity depending on animal group. High nectar producing plant richness for butterflies occurs throughout the north west and Hebrides (Fig. 5.6a) but bee nectar plants become more restricted to coastal habitats (Fig. 5.6b).

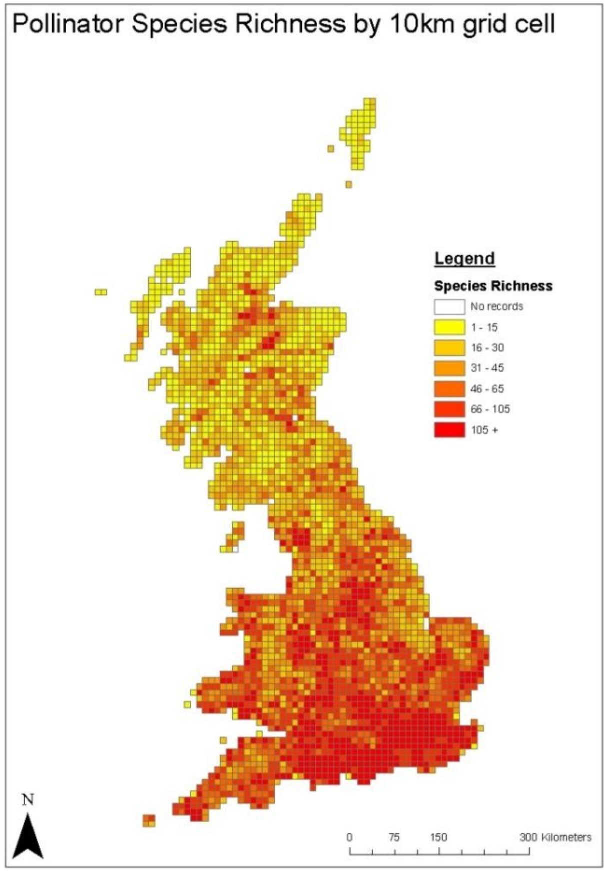
[33] Source: Biological Records Centre, CEH (www.brc.ac.uk)

Figure 5.4: a) Bumblee and solitary bee species-richness (post-1970)[34], b) Total pollinator insect species-richness (butterflies, bees and hoverflies).

a)



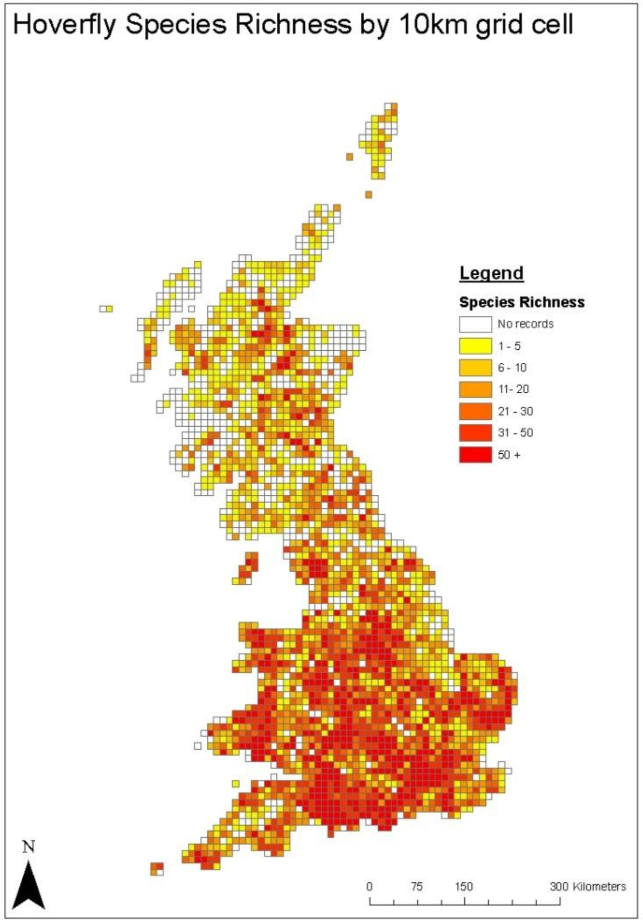
b)



[34] Records of bee species collected by the Bees, Wasps & Ants Recording Society (BWARS) (www.bwars.com/about_BWARS.htm) and held by the Biological Records Centre at CEH Wallingford.

Figure 5.5: a) Hoverfly species-richness post-1970, b) Butterfly species-richness (1995-2004).

a)



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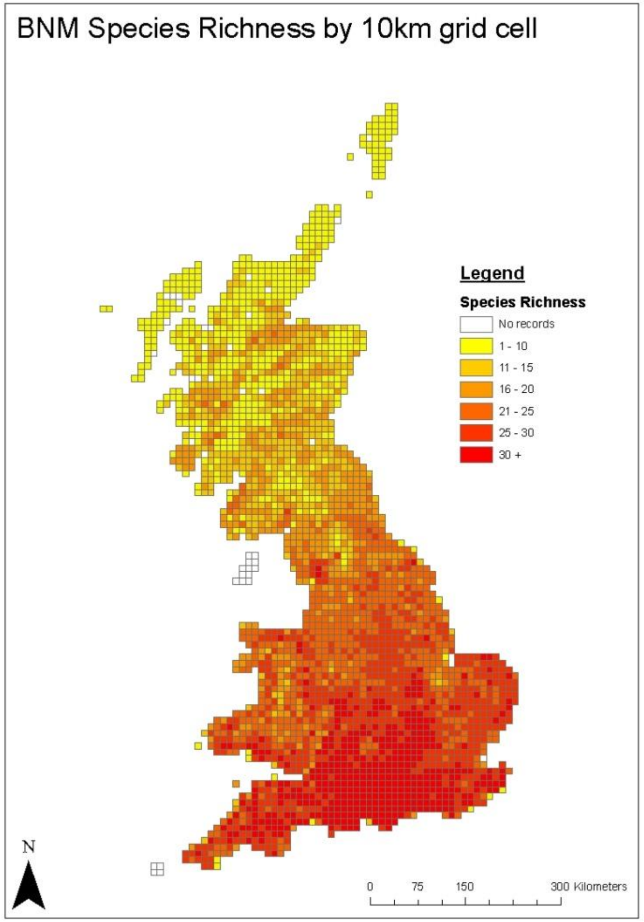
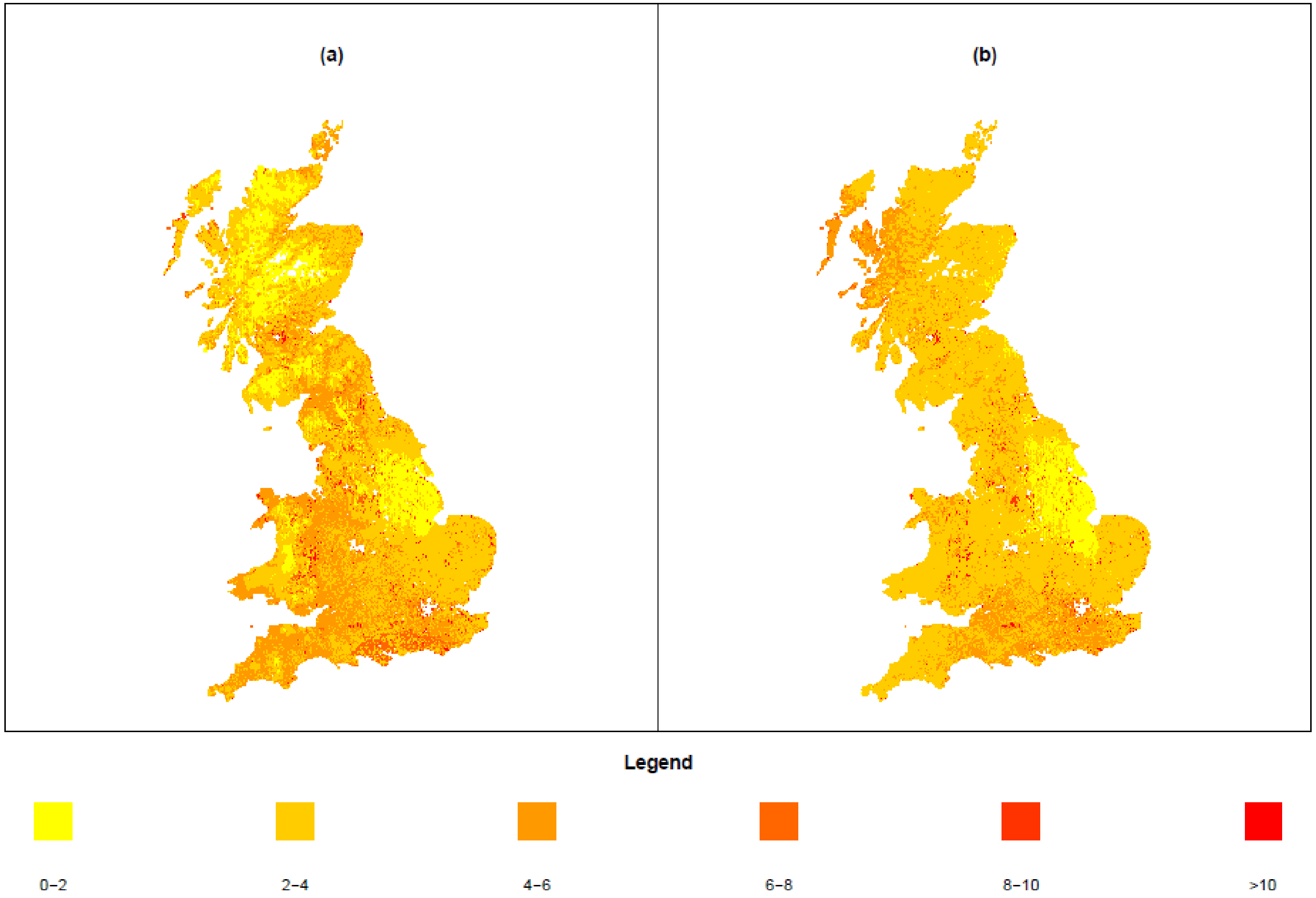


Figure 5.6: Predicted mean bee (a) and butterfly (b) nectar plant richness per 4m^2 plot in 2007 averaged over the Broad Habitats in each 1km square. Predictions generated from a Generalised Additive Mixed Model that included Broad Habitat, climate, habitat patch size, length of linear features in each 1km square and ELS status in 2007 as explanatory variables (see section 5.8 and Appendix 5.2).



5.7 Explaining change in nectar plant diversity between 1990 and 2007

Selecting expected drivers and environmental correlates of change

Evidence for impacts on nectar plant abundance

In Britain, previous analyses have shown that nectar-providing plant species have declined disproportionately across Britain compared to other native species. Carvell *et al.* (2006a) analysed changes in occupancy of 10km grid squares between the periods 1930-'69 and 1987-'99, and in frequency within vegetation plots between the earliest Countryside Survey in 1978 and the 1998 survey. Results showed clear evidence of greater declines in nectar species than other native plants with 76% of the listed nectar plant species showing significant declines in CS vegetation plots in the 20 year period. These changes seemed likely to capture the culmination of the post-war period of large-scale habitat loss due to agricultural intensification and urban development in Britain (Goulson *et al.* 2005; Williams 2005; Robinson and Sutherland 2002). Bee species diversity in Britain is also known to have declined during the post-war period (Williams 1982) and a recent analysis of correlated temporal changes in pollinating insects and nectar plants showed that patterns of change in abundance in Britain and the Netherlands were consistent with a mechanistic relationshsip linking the decline in bees to the decline in plants (Beismeijer *et al.* 2006). However, such consistent spatial patterns do not necessarily help in generalising the direction of the causal relationship. Do declines in pollinating insects lead to declines in wild plants, particularly obligate outcrossers? Or does declining abundance of nectar sources cause a decline in insect populations? Evidence exists for the operation of causal chains in both directions but it is an open question as to which mechanism prevails at the large scale across Britain and other agriculturally managed temperate ecosystems.

Irrespective of the relative importance of each mechanism, a common trait-related pattern can be seen to charaterise changes in the fortunes of both insect and plant groups in Britain and this helps to identify likely drivers of change in pollinator plants as well as identifying vulnerable plant types. Observed reductions in bee and nectar plant diversity have happened in parallel with the non-random selection of traits among 'winners' or 'losers'. Butterflies, bees and plants that have prospered in the last 50 years have tended to be generalists that can exploit a range of resources and conditions. Species that are specialised to unproductive, less disturbed conditions have disproprtionately suffered (Smart *et al.* 2000; 2006b; Carré *et al.* 2009; Beismeijer *et al.* 2006).

Impacts of positive management

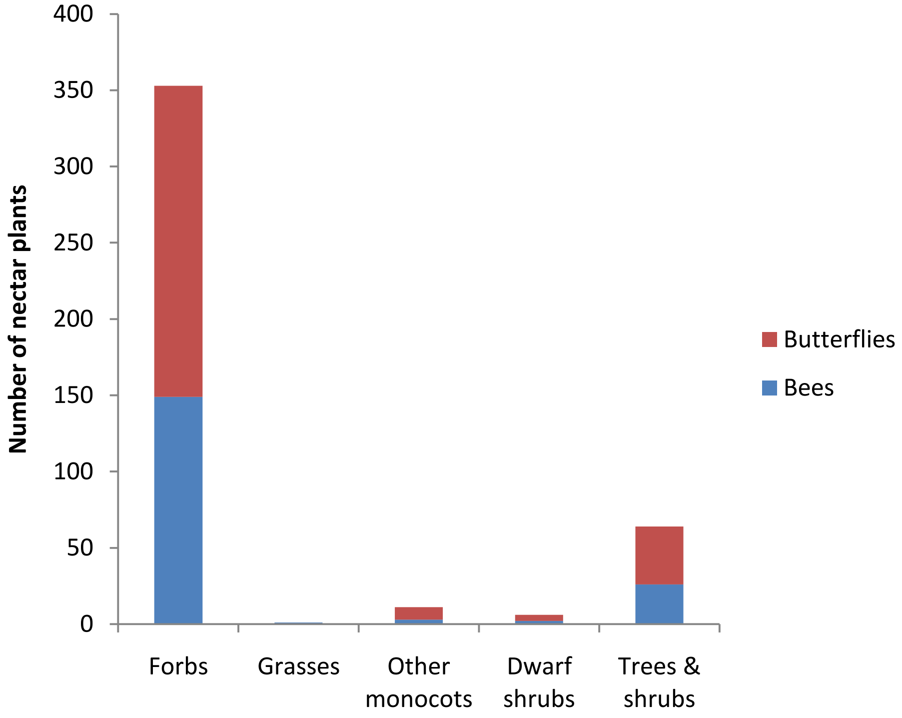
The opportunities provided by increasing abundance of non-native plant species, greater suburban niche space and increases in mass-flowering crops are not likely to offset the large-scale declines in specialised native insects and plants (Carvell *et al.* 2006a; Carré *et al.* 2009). In the countryside, practical extensification measures can, however, boost wild pollinator insect and nectar plant populations increasing farm-scale biodiversity and delivery of pollination services to adjacent crops (Pywell *et al.* 2005; Carvell 2002; Carvell *et al.* 2006b). Whether such benefits are realised at the large scale depends upon the geographic penetration of such measures. Previous schemes were regionally based or competitively applied so that only targeted areas of responsive or high biodiversity agricutural land received funding. The new Entry Level Scheme in England plus similar schemes in Wales and Scotland offer scope for wider national uptake of farm-scale measures that may yield detectable signals in surveillance data if sufficiently widespread. For example, the impact of setaside has almost certainly been detected in Countryside Survey plot data in the past (Smart *et al.* 2005; Carey *et al.* 2008).

Attributing signals of change to positive management schemes depends upon the strength of the signal in the surveillance data. This is governed by the time since management divergence was implemented and on the responsiveness of the vegetation. The chances of detecting any signals also crucially depend on the availability of spatially precise information on where agri-environment options were applied. We applied a dataset for English farmland where polygons were either under ELS options or not. Since many margin options are rotational in nature whilst the longest time that management could have been in place is two seasons prior to the 2007 survey, signals of scheme impact were not expected.

Hypothesising links between change in nectar plant diversity and potential drivers of change

Possible drivers of change in mean nectar plant richness in Britan include the range of global change phenomena (Sala *et al.* 2001) already examined in Chapter 4. These comprise recent climate warming (Morecroft *et al.* 2009), atmospheric pollution (Smart *et al.* 2004; Maskell *et al.* 2010) and land-use (Carvell *et al.* 2006a; Heard *et al.* 2007; Smart *et al.* 2006a; Firbank *et al.* 2008). Hypothesised impacts of these drivers were expected to follow those mechanisms and directions of change already considered for indicators of appropriate diversity in Chapter 4. This is because the distribution of growth forms for nectar plants is similar to that for CSM Indicators. In particular the vulnerable 'forb' group is strongly represented (Fig. 5.7).

Figure 5.7: Distribution of growth forms among nectar plant species for butterflies and bees (bumblebees and solitary bees).



Both bees and butterflies depend upon a much higher proportion of forbs than other growth forms. Evidence indicates that forbs, especially small stress-tolerators have disproportionately suffered from recent human activity across Britain whereas grasses and woody species have often increased (Smart *et al.* 2005; Hodgson *et al.* 2005; Walker *et al.* 2009; Preston *et al.* 2002). Even so the likely impact of drivers will depend upon the species composition of the habitat type analysed[35]. Later-successional vegetation or habitat types characteristically poor in forbs reflecting either high or low productivity would be expected to show less responsiveness to succession and eutrophication (Foster 2001). If not dispersal limited (Marrs *et al.* 1996), unproductive vegetation types could increase or decrease in nectar plant richness following increased nutrient status whilst productive vegetation types would be expected to increase in richness if productivity decreased. In general, secondary succession would be expected to reduce nectar plant richness in plots if this resulted in increased woody cover and shading of mid-successional assemblages (Smart *et al.* 2006a). However, if species-poor, frequently disturbed plots experienced reduced disturbance, then small increases in nectar plant richness could occur and these might be more likely to involve initial increases in tall forbs, trees and shrubs. In general, responses to succession and eutrophication are likely to depend on the starting ecological conditions of the habitat concerned (Smart *et al.* 2006b).

[35] See Appendix 5.1 for nectar plant frequency by Broad Habitat.

Drivers and their best available explanatory variables are listed in Table 5.2 alongwith the pressures and processes associated with each driver. Three additional types of analysis were also carried out:

The effects of sheep grazing in upland Broad Habitats

Correlative relationships were tested between nectar plant diversity change and sheep density in upland Countryside Survey squares (defined as being in Environmental Zones 3, 5 and 6 – see Haines-Young *et al.* (2003). Two explanatory variables were applied; the linear slope of change in sheep numbers in each AgCENSUS 4km^2 between 1969 and 2000, and the total number of sheep in the 4km^2 in 2000. Analyses were carried out by upland Broad Habitat and by plot-type discriminating small habitat patches (Targeted Plots) and larger areas of unenclosed land (Main and Unenclosed Plots).

Extreme weather – after effects of the October 1987 storm

Species richness of plants and animals is known to have increased in many of the woodlands impacted by canopy destruction in the 1987 storm (Kirby *et al.* 2005; Whitbread 1993). This is an expected response to gap creation which would favour diversification and natural recolonisation. All plots in Broadleaved, Mixed and Yew Woodland (Main, Targeted and Unenclosed) were selected from within the storm track (n=28 1km squares and 204 plots) and compared with the same plot types in woodland in the lowland zone 1 outside the storm track (n=54 1km squares and 312 plots).

Early signals of the Entry Level Scheme

Using a polygon-level spatial coverage for England, Countryside Survey plots were selected that were either in or out of an ELS agreement in 2007. The details of the scheme were examined (Natural England 2008) and plots were selected and divided into four groups according to Broad Habitats and landscape locations likely to have been affected by similar options as follows:

1. Field margins and hedges adjacent to Improved and Neutral Grassland
2. Streamside Plots associated with Improved and Neutral Grassland
3. Area and Linear Plots associated with Arable and Horticulture
4. Area plots (Main, Targeted and Unenclosed) in Acid Grassland, Calcareous Grassland, Bog, Dwarf Shrub Heath and Fen, Marsh and Swamp.

Since the scheme opened in 2005, the analysis focused just on differences between plots in 1998 and 2007.

Table 5.2 Possible drivers of change in nectar plant diversity in Countryside Survey plots across Britain between 1990, 1998 and 2007. See Table 4.2, Chapter 4 for notes on mechanisms and references specific to each Broad Habitat. Unless otherwise stated, two analyses were carried out for each Broad Habitat. One analysis focused on patterns in larger areas of habitat and unenclosed land (Main and Unenclosed Plots) and a second analysis examined patterns within small habitat fragments located within larger areas of each Broad Habitat (Targeted Plots). The Boundary and Linear Features Broad Habitat was defined by plots on field boundaries, in hedges and on road verges. The Rivers and Streams Broad Habitat was defined by Streamside Plots.

Driver	Explanatory variable	Process/pressure
Atmospheric pollutant deposition	Change in sulphur deposition between 1971 and 2005[36]	Recovery from acidification
"	Total nitrogen deposition averaged for 2004-'06[37]	Eutrophication/ acidification
Climate change	Linear slope of change in minimum January temperature from 1980 to 2005	Altered net primary production with non-random impacts on the plant trait pool
"	Linear slope of change in annual precipitation from 1980 to 2005	"
Extreme weather	Plots in or outside track of October 1987 storm	Successional response to large-scale canopy destruction
Land use	% Improved Grassland and Arable and Horticulture in each 1km square	Effects of intensive land-use; species pool dominated by nutreint-demanding species; exposure to nutrient surpluses; direct impact of intensive production methods
"	Sheep numbers in 2000	Agricultural intensification in upland Britain
"	Standardised linear slope of change in sheep numbers between 1969 and 2000	"
Extensification	Land parcels under Entry Level Scheme agreement or not (England only)	Reduced inputs; no overgrazing; direct establishment of diverse seed mixtures including nectar plants.

[36] For SOy there is no separation into wet and dry but estimates for forest, grassland and moor were attached to each plot subset reflecting BH identity in 1990.
[37] N deposition was divided into reduced and oxidised and estimates for forest, grassland and moor were attached to each plot subset reflecting BH identity in 1990. Note that N deposition was not applied to Improved Grassland or Arable since atmospheric inputs fall short of agricultural inputs.

Explantory variables in Table 5.2 were all measured at grid square scales ($4km^2$ for AgCENSUS data and $5km^2$ for climate and pollutant deposition data). These variables could not therefore explain between-plot variation within each 1km square. A number of additional plot-level variables were used as correlates of processes that changed the species composition over time in ways that could have impacted nectar plant richness. These were as follows:

1. Mean change in DCA[38] axis 1 score between 1990, 1998 and 2007. The first axis in the ordination of CS data is interpreted as a substrate productivity axis so changes in score indicate shifts towards more or less nutrient-rich conditions (Bunce *et al.* 1999a). Changes in the score could reflect a number of linked drivers such as climate warming and changes in rainfall, as well as land management and atmospheric nitrogen deposition. Apart from presence in or outside ELS agreement, explanatory variable data that could explain changes in axis score in terms of these drivers, is lacking at the plot scale.
2. Mean change in DCA axis 2 score between 1990, 1998 and 2007. The second axis in the ordination of CS data is interpreted as a successional axis from short, open, disturbed communities through to woodland. Changes in score therefore indicate shifts towards more or less frequently disturbed vegetation.
3. Change in cover of trees and shrubs (excluding dwarf shrubs). This is a simple and transparent indicator of successional change at the plot level.

Analytical methods

The response variable in all tests was the count of nectar plants in each plot in 1990, 1998 and 2007. Data of this sort has several properties that need to be taken into account in the analysis. Values can never be less than zero whilst species-rich plots have more species to lose. Higher mean richness values also have proportionately higher variance. These properties are accounted for in the analytical model by applying a Poisson distribution with an automatic correction for greater or lesser variability than expected by this distribution. This is broadly equivalent to analysing natural logs of the species counts using a normal distribution.

The structure of the sampling design means that a mixed model is appropriate as used in other chapters and described in Maskell *et al.* (2010). The analysis of Entry Level Stewardship (ELS) agreement land required comparison of data from 1998 and 2007 only. Hence, repeat plot counts were reduced to one set of differences and the data analysed with a normal error distribution.

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[38] Detrended Correspondence Analysis (DCA) is an ordination technique used to quantify and convey patterns of species compositional similarity in such a way that they can be interpreted as ecological gradients. The technique was used to identify the key gradients (fertility and shade/disturbance) in CS data that were subsequently used to group vegetation plots into CVS classes and Aggregate Classes (see Bunce *et al.*1999a)

Hypothesis tests

The year of survey was paramaterised with 1990 being coded 0. This means that the intercept term estimates the mean richness in the first survey and interpretation of additional terms for the time effect and other explanatory variables are simplified (Singer 1998). To determine whether an explanatory variable could explain significant change in nectar plant richness over the three surveys, time and each explanatory variable were entered as main effects and then the interaction between them tested for significance. A significant overall effect of time indicates whether nectar plant richness increased or decreased over the 17 year period and essentially repeats the tests reported in section 5.5. A significant interaction with the explanatory variables listed above indicates that change in time was conditional on the value of the explanatory variable. For example, mean nectar plant richness may have decreased overall on streamsides but may have decreased more where the vegetation became more overgrown or more fertile.

In all cases, partial tests were carried out. This means that the presence of a significant interaction was tested in the presence of other interaction terms. This is a conservative approach but guards against falsely interpreting an explanatory variable as a significant possible driver of change because it is highly correlated with gradients of other drivers that could also have plausibly generated the observed effect (Maskell *et al.* 2010). In addition, each 1km square rather than each individual plot was specified as a repeated subject. This is again a somewhat conservative approach but guards against regression to the mean effects caused by relocation error switching extreme plots back toward the average between surveys (Palmer 1993).

In all cases the Broad Habitat membership of each plot was taken as that applying in 1990.

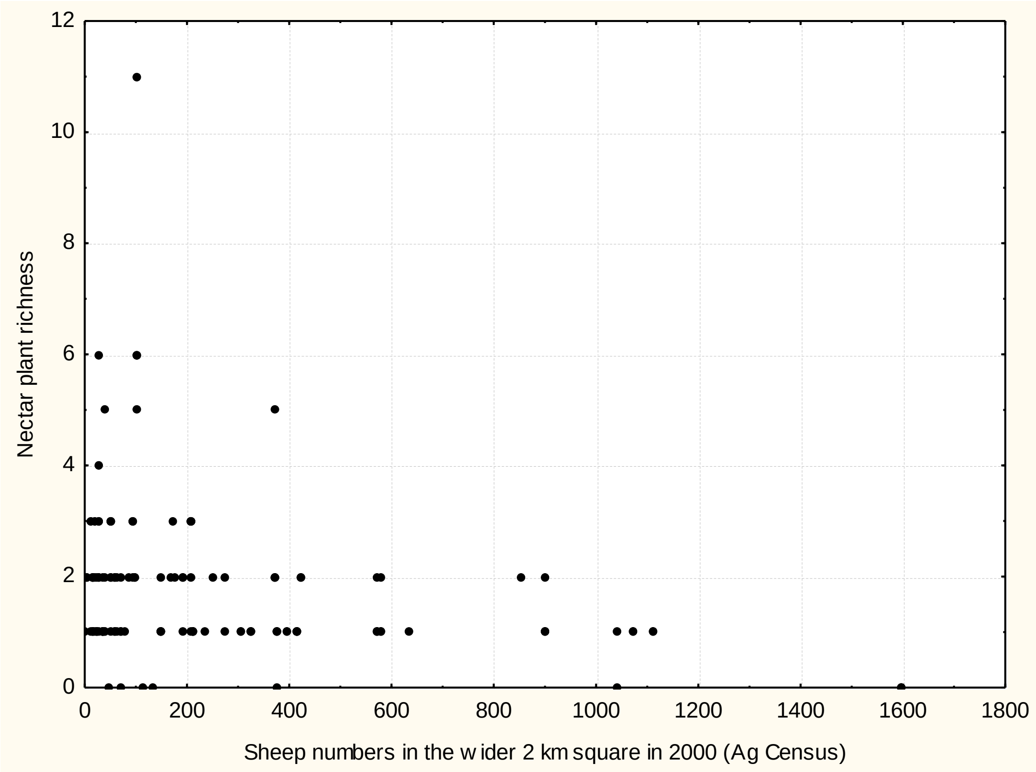
Results

Significant drivers of change in nectar plant diversity

In no instance were any of the grid square-level explanatory variables significant in explaining change in nectar plant richness over time. No effect of the October 1987 storm was detected nor any difference between plots in polygons under Entry Level Stewardship (ELS) agreement in England.

Sheep numbers estimated for upland 1km squares in 2000 had a negative spatial correlation with nectar plant richness but the lack of any significant interaction with temporal change in richness suggests that this pattern was already visible by 1990. The strongest relationship with sheep numbers was found for Dwarf Shrub Heath (Fig. 5.8) with significant but weaker relationships in Fen, Marsh and Swamp and Bog (Table 5.3).

Figure 5.8: Nectar plant richness per Main Plot in upland Dwarf Shrub Heath in Britain in 1990 versus sheep numbers in the wider 2x2 km square in 2000.



The lack of significance of drivers measured by variables at the grid square scale suggests either that these variables were highly intercorrelated or that average changes in nectar plant richness between squares were not related to these variables. Intercorrelation is a possibility but initial graphical exploration of the dataset for each analysis was carried out and highly correlated pairs of variables reduced to just one variable. However, this does not completely remove the chances of lack of significance in the partial tests because of shared variability. Since square level variables were significant in analyses of appropriate diversity (see Chapter 4) it is likely that changes in nectar plant richness are simply not well explained by the variables selected. This is not too surprising. For example, Maskell *et al.* (2010) found that *spatial* gradients in species richness were often explained by square-level variables but analysis of CS data for the Review of Transboundary Air Pollution (ROTAP) (Fowler *et al.* in prep) showed that while spatial gradients were strongly detectable, far fewer significant relationships emerged between *temporal* change and square-level predictors. In the analysis of change in nectar plant richness, plot-level variables were much more important but they are not independent of the species composition of each plot and so cannot directly point to specific drivers such as land-use, atmospheric pollution or climate change.

Significant conditional effects of explanatory variables on change in nectar plant richness since 1990 were confined to plot-level changes in species composition along inferred axes of productivity and successional stage. In three Broad Habitats, change in cover of trees and shrubs was also significant (Table 5.3). The strongest cross-habitat pattern was a negative effect of successional change on nectar plant richness; where the species assemblage had on average moved to a taller, less disturbed vegetation type then nectar plant richness declined to a greater extent. With the exception of Acid Grassland, this relationship was seen only in later successional vegetation already characterised by woody canopy cover and where further succession was likely to have coincided with greater shade at ground level. The positive effect of succession on nectar plant species richness in Boundary and Linear Features was an exception to this pattern (Table 5.3).

Changes along the substrate productivity axis were also correlated with changes in nectar plant diversity in Coniferous Woodlands, Bracken and Acid Grassland, changes toward assemblages associated with lower productivity were correlated with reduced nectar plant richness whilst in Neutral Grasslands and Arable and Horticulture, a change toward less productive assemblages was linked to greater nectar plant diversity (Table 5.3).

Are correlations consistent with expectation?

The most important features of the results are the correlations between nectar plant diversity change with successional change and substrate productivity change. Movement along these two gradients is itself linked by the fact that greater shade favours stress-tolerators, which also favour low productivity (Kirby *et al.* 2005; Smart *et al.* 2006a). However, sufficient variation must have been uniquely attributable to each gradient in those situations where both were significant in the same partial test (Table 5.3). Only in Boundary and Linear Features was successional change linked to increasing nectar plant diversity and this may reflect the responsiveness of early to mid-successional vegetation to relaxed disturbance and where the end-point by 2007 was not high cover of trees or shrubs. In Broadleaved and Coniferous Woodland and Rivers and Streams, greater succession tended to exacerbate loss of nectar plant diversity. In these habitats, succession is more likely to result in gap closure and suppression of herb richness. In Streamside Plots in particular, high initial herb richness increases the chances of a larger reduction in response to shading, because there are more species to lose (Smart *et al.* 2006a). In addition, streamsides in lowland Britain are also known to have become much more tree and shrub dominated over the past 30 years (Carey *et al.* 2008).

The response to changes along the productivity gradient (DCA axis 1) differed depending upon Broad Habitat type. In Acid Grassland and Conifereous Woodland reductions in nectar plant diversity were correlated with a shift to less productive plant assemblages. In Arable and Neutral Grassland, reduced productivity was associated with increased nectar plant diversity. These responses would be expected given the different mean productivity of the vegetation types (Fig. 5.9). Arable and Neutral Grassland are situated toward the productive right side of the diversity-productivity curve (Smart *et al.* 2003). Given the relationship with nectar plant richness, which mirrors that of total species richness, a reduction in productivity should be associated with an increase in nectar plant richness in these habitats. Acid Grassland and Coniferous Woodland are situated on the mid-point and to the left of the curve so that a reduction in productivity ought to be associated with reduced richness. The fact that consistent changes have been detected in CS data is interesting. These patterns have been shown before in CS data for total species richness in response to apparent setaside uptake on arable land between 1990 and 1998 (Smart *et al.* 2003) and have been demonstrated in other ecosystems (Wright and Jones 2004). However, lack of a response is also typical and to be expected in agricultural landscapes where species pools have been depleted of poorly dispersing perennials (Marrs *et al.* 1996; Kleijn and Sutherland 2003; Grime 2002).

Figure 5.9: Inferred substrate productivity and bee nectar plant diversity across British Broad Habitats. Data points are ordination axis scores and nectar plant counts for 1728 Main plots in 1990. The arrows show how the directions of change in nectar plant richness that result from movement toward less productive species assemblages depend critically on starting point along the gradient.

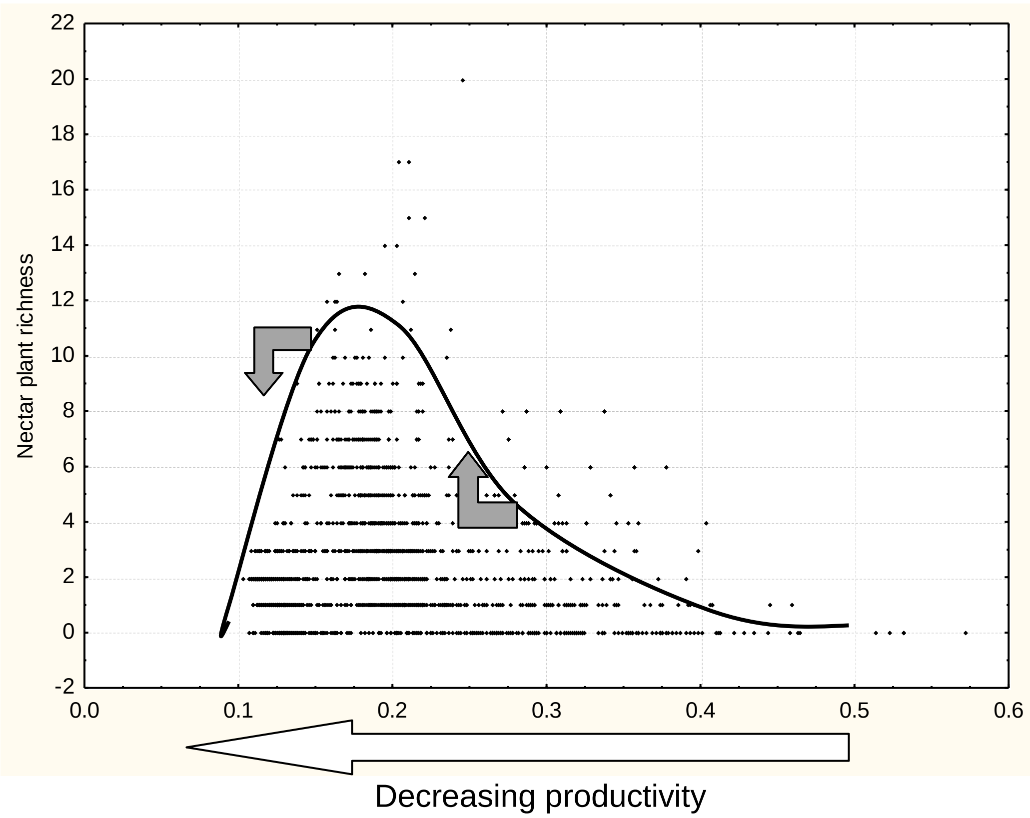


Table 5.3: Significant correlated drivers of change in nectar plant richness in British Broad Habitats between 1990, 1998 and 2007 based on analysis of a) larger areas of common habitat represented by Main and Unenclosed Plots or Linear Plots for linear habitats and, b) small habitat patches represented by the Targeted Plots[39]

a) Larger areas of common habitat and unenclosed land.

Broad Habitat	Significant correlative effects	Interpretation
Arable and Horticulture	Mean change along DCA axis 1	A shift toward less productive species assemblages promoted increased nectar plant richness on arable land.
Boundary and Linear Features	Mean change along DCA axis 2	Successional change promotes increased nectar plant richness along road verges, hedge bases and field boundaries.
Coniferous Woodland	Mean change along DCA axes 1 and 2. Change in tree and shub cover.	Successional change, involving an increase in tree cover, exacerbates loss of nectar plants as does a shift toward less productive species assemblages
Acid Grassland	Mean change along DCA axes 1 and 2.	Successional change exacerbates loss of nectar plants as does a shift toward less productive species assemblages.
Bracken	Mean change along DCA axis 1.	A shift toward less productive species assemblages exacerbates loss of nectar plants.
Dwarf Shrub Heath	Sheep numbers in 2000	No interaction with time suggesting that the grazing related gradient was in place by 1990. Fewer nectar plants present at higher sheep numbers in uplands.
Fen, Marsh and Swamp	Sheep numbers in 2000	No interaction with time suggesting that the grazing related gradient was in place by 1990. Fewer nectar plants present at higher sheep numbers in uplands but a much weaker effect than for Dwarf Shrub Heath.
Bog	Sheep numbers in 2000	No interaction with time suggesting that grazing related gradient was in place by 1990. Fewer nectar plants present at higher sheep numbers in uplands but a much

[39] These patches are diverse in species composition but typically pick out atypically rich or distinctive assemblages not common enough in each 1km square to have been sampled by the random plots analysed above (Smart *et al* 2006a). Examples include small flushes, bog pools, hydroseres around waterbodies, species rich grassland fragments in field corners or on steep slopes, coastal assemblages, species rich weed communities, distinctive woodland gaps and rides. A common theme among such patches is their early to mid-successional status. This means that without appropriate disturbance species composition may well undergo directional change (Smart *et al* 2003). In Countryside Survey these habitat patches are represented by the targeted Y plots (Carey *et al* 2008).

		weaker effect than for Dwarf Shrub Heath.
Rivers and Streams	Mean change along DCA axes 1 and 2. Change in tree and shub cover.	Successional change, involving an increase in tree cover, exacerbates loss of nectar plants as does a shift toward less productive species assemblages.
Broadleaved, Mixed and Yew Woodland	None	
Improved Grassland	None	
Neutral Grassland	None	
Calcareous Grassland	None	

b) Small habitat patches.

Broad Habitat	Sig correlative effects	Interpretation
Neutral Grassland	Mean change along DCA axis 1	A shift toward less productive species assemblages promoted increased nectar plant richness in semi-improved and unimproved grasslands.
Broadleaved, Mixed and Yew Woodland	Mean change along DCA axis 2	Successional change exacerbates loss of nectar plants in broadleaved woodland.
Coniferous Woodland	Mean change along DCA axis 1.	A shift toward less productive species assemblages exacerbates loss of nectar plants.
Calcareous Grassland	Change in tree and shrub cover	Increasing woody cover was correlated with a loss of nectar plants.
Acid Grassland	None	
Bracken		
Dwarf Shrub Heath	Sheep numbers in 2000	No interaction with time suggesting that grazing related gradient was in place by 1990. Fewer nectar plants present at higher sheep numbers in uplands but a much weaker effect than for area plots.
Fen, Marsh and Swamp	None	
Bog	None	
Arable and Horticulture	None	
Improved Grassland	None	

5.8 Discussion

The significance of recent changes in nectar plant richness for pollination service delivery

It is possible that the changes in mean nectar plant richness we have detected between 1990 and 2007 may represent a small-magnitude signal of the culmination of post-war agricultural intensification. It might also be the case that these recent changes track a new phase of more modest reduction in diversity driven by more subtle drivers relative to mid-20th century habitat destruction, such as climate change, atmospheric pollution and a relaxation in management intensity in woodlands and on linear features and streamsides (Smart *et al.* 2007; Kirby *et al.* 2005). Changes during the last 17 years may also have been small because major episodes of impoverishment in the wider countryside had already occurred so there are now fewer species left to lose. The more substantial reductions in bee and butterfly nectar plants detected in CS data for 1978 through to 1990 and 1998 (Carvell *et al.* 2006a; Smart *et al.* 2000) and in 10km plant occupancy data (Carvell *et al.* 2006a; Beismeijer *et al.* 2006) are perhaps more likely to reflect impacts associated with the end point of post-war mechanisation of agriculture and the drive to increase food production.

The mean changes in nectar plant richness detected in CS data may not have severely depleted the delivery of pollination services because reductions in nectar plant diversity had already happened by the time of the 1990 survey. This does not rule out locally large changes in nectar plant richness that are not adequately represented by Broad Habitat averages. A better understanding of the functional significance of observed changes in nectar plant diversity could be achieved by calibrating changes in Countryside Survey plots against increases or decreases in pollinator insect diversity and visitation that result from experimentally-driven changes in the abundance of nectar plants.

The largest reductions in nectar plant richness occurred in small semi-natural habitat patches. For example the mean richness in Neutral Grassland Targeted Plots declined by 2007 to less than the 1990 mean for Improved Grassland (Fig. 5.3). This is important because we would expect Improved Grassland to be less species rich than Neutral Grassland. Hence, this loss represents a loss of species diversity which may be changing the character of the Broad Habitat and its ability to support nectar plants and polliniating insects.

Overall though, changes in mean richness were small and not enough to shift the mean for one Broad Habitat further toward the values typical of a less species rich habitat (see section 5.5). In addition, prospects for increasing nectar plant diversity on farmland are currently good since agri-environment options in England and Wales offer scope for low level but geographically widespread benefits to pollination services (Carvell *et al.* 2006b). Whilst some options include direct sowing of beneficial seed mixtures, in general there is a reliance on regional species pools retaining enough diversity to be responsive to new opportunities for establishment where these agri-environment schemes have been implemented. The role of refuge features such as small remnant habitat patches and linear features as well as overall habitat diversity and the resilience of seedbanks may be crucial in this respect (Heard *et al.* 2007).

5.9 Development and application of a statistical model predicting nectar plant diversity across Great Britain

Introduction

The previous sections of this chapter quantified recent change and current status of nectar plant diversity across British Broad Habitats and then sought to explain change over time in terms of plausible driving variables. The understanding gained from this work can be combined with ecological knowledge to hypothesise those factors that ought to explain spatial differences in nectar plant diversity from place to place. In this section we turn this knowledge into a statistical model that can be used to predict nectar plant diversity in small patches ($4m^2$) of vegetation at the large scale. Since the datasets available to build such a model come from vegetation plots across Britain, model predictions can be made at the detailed level of mean values per habitat patch.

Depending upon the explanatory power and identity of the predictors in the model, and the availability of predictor data for new regions, it should be possible to test scenarios of the impact of future environmental change on potential nectar plant diversity. This section finishes with a demonstration of the capability of the model to predict potential change in the English uplands given a published scenario of expected land cover change driven by policy incentives to farmers.

Methods

Generalised Additive Mixed Models (GAMMs) were used to predict nectar plant diversity for bees and butterflies in terms of a range of ecologically plausible explanatory variables. The modelling technique has already been fully described in Chapter 4, Appendix 4.3.

Table 5.4: List of drivers of spatial diversity and explanatory variables included in models

Driver of spatial diversity patterns	Potential Explanatory variables	Source
Environmental factor	Altitude	Ordnance Survey
Environmental factor	Slope of plot	Measured in plot in CS2007
Environmental factor	Aspect of plot	Measured in plot in CS2007
Land use	Shade class of plot	Measured in plot in CS2007
Land use	Canopy height of vegetation	Measured in plot in CS2007
Land use	Broad Habitat type of plot	Mapped in CS in 2007
Land use	% of woody cover in plot	Calculated from species abundance data
Land use	Coincidence with agri-environment scheme	Polygon-level assignment of each plot for England only in 2007 (source Natural England)
Land use	Total length of linear features in 1km square	Mapped in each CS 1km square in 2007
Land use/ environmental factor	Total area of polygon containing plot[40]	Mapped in each CS 1km square in 2007
Climate	Mean annual temperature	UKCIP long-term averages (1969 to 2000) (5x5km)
Climate	Annual rainfall	UKCIP long-term averages (1969 to 2000) (5x5km)
Climate	Long-term mean daily sunshine hours	UKCIP long-term averages (1969 to 2000) (5x5km)
Climate	Mean daily % cloud cover	UKCIP long-term averages (1969 to 2000) (5x5km)
Atmospheric pollution	Total nitrogen deposition	CBED averages for 2004,'05 and '06 (5x5km)

Datasets

The response variables to be predicted were counts of nectar plants for bees (bumblebees and solitary bees combined) and butterflies - the same data as analysed in previous sections of this chapter. Explanatory variables were initially listed based on the plausibility of their likely effect on richness of nectar plants in Countryside Survey plots. Table 5.4 shows a full list of

[40] Where polygons were bisected by the edge of each 1km square, the complete area of the polygon was estimated using Bayesian inference given the known distribution of areas of polygons falling entirely within the 1km squares.

variables that were available for inclusion in the models. The number, identity and scale of the variables chosen reflect data availability as well as ecological justification. It is possible that more variation in the CS data could be explained with more finely resolved data on management impacts (including agri-environment scheme status) (Table 5.4).

Scenario testing

The best fitting models were used to predict nectar plant diversity given adjustments to the selected model covariates. Scenarios of changing land-cover were applied to estimate potential impacts on nectar plant diversity. We adopted a scenario based on the Defra (2006) report into the future reward structure in the UK uplands. The report suggested three policy options for changes in the UK uplands and sought opinions as to the most beneficial policy scenario for “rewarding sustainable land management and the provision of public benefits in the uplands” (Defra 2006). The specific policy scenario chosen was the “environment only” option. Under this scenario, rewards are in the form of Entry Level Scheme payments with no Common Agricultural Policy-style subsidy given purely on the basis of area owned. This scenario was thought likely to drive local abandonment of land for agricultural purposes because of the lack of return on marginal land. Increased afforestation was anticipated but also more focused positive management on smaller areas likely to be responsive to scheme options (Hanley and Colombo 2009). Under the same scenario, the upland Entry Level Stewardship scheme is also thought likely to result in increased arable land. The Cumulus report (Cumulus Consultants Ltd., 2005), commissioned by Defra and cited in the consultation document (Defra 2006), sets out how physical attributes of the landscape may change under each of the proposed scenarios.

Having selected our scenario, we used the results of the Cumulus report to apply expected changes in proportional land cover in each 1km square. The covariate of interest and that which is subject to change in the scenario test was Broad Habitat, as this corresponds with the landscape attributes quoted in the Cumulus report. From such changes, model predictions could be made and the responses of potential nectar plant diversity under the scenario could be analysed. Table 5.5 is an extract from the Cumulus report, which shows the expected percentage stock change of various landscape attributes under the “environment only” policy scenario.

Table 5.5: Extract from Cumulus Consultants Ltd. (2005) report showing changes to landscape attributed under Defra's "environment only" policy scenario.

Upland Landscape Attribute	Change under Policy Scenario
Heather moorland and bog	+5%
Improved grassland	0%
Rough grassland	-3%
Hay meadow	0%
Bracken dominated	+3%
Gorse dominated	+2%
Arable (and set aside / fallow)	-4%
Broadleaf and mixed woodland	+6%
Coniferous woodland	-6%
Field boundaries	+10%
Cultural heritage	No change

For each Broad Habitat expected to increase in area, we selected donor areas from the Broad Habitat types expected to decline in area given rules that we arbitrarily constructed concerning the likelihood of one habitat being replaced by another (Table 5.6).

Table 5.6: Landscape attributes with an increase in stock, from Table 5.5, together with the landscapes that may have changed to contribute to that increase.

Attribute	Change	Habitat most likely to be gained from…
Heather moorland and bog	+5%	Coniferous woodland and rough grassland
Bracken dominated	+3%	Rough grassland and Arable (and set aside / fallow)
Gorse dominated	+2%	Rough grassland
Broadleaf and mixed woodland	+6%	Coniferous woodland and rough grassland
Field boundaries	+10%	Rough grassland and Arable (and set aside / fallow)

Under the scenario, the changes in attributes shown in Table5.6 only applied to Severely Disadvantaged Areas in England. We therefore restricted the scenario testing to data observed from 1 km squares falling within these areas based on Land Cover Map 2000. To test the scenario and its affects on the response, we could have chosen a sample of our observations, with size corresponding to the appropriate level of stock change, change their landscape type variable in accordance with Table 5.6 and use this new covariate data to predict the response. However, rather than implementing this change at a single sample of sites, we used a bootstrap procedure to gain better insight into the average implications the changes have on the response variable. To do this we used the following algorithm:

1. Select a habitat type listed in the first column of Table 5.6. Denote this T.
2. Note the habitat type that contributes to the increase in T – this can be read off from the third column in Table 5.6. Denote this habitat S.
3. Randomly sample *x* polygons with habitat type S, where *x* is the number of polygons corresponding to the area increase in T.
4. For the *x* sampled polygons, change their habitat type from S to T.
5. Repeat steps 1 - 4 for each habitat type listed in the first column in Table 5.6.
6. Using each fitted model, obtain a set of predictions using the new information on habitat type.
7. Store the set of predictions.
8. Repeat steps 1 – 7, 1000 times to obtain 1000 predictions at every 1km square in the data set.
9. Calculate the mean and standard error of the 1000 predictions for each 1km square.

Where an increase in a habitat attribute was likely to be gained from two or more other habitat attributes (Table 5.6), the contribution to the increase was assumed to be equal. For example, Bracken dominated habitats are predicted to increase by 3%, which may correspond to an increase of 100 sites. This increase is likely to be detrimental to arable land and rough grassland. Under the equal contribution assumption, 50 sites that were designated as rough grassland are changed to Bracken and 50 sites that were designated arable land are changed to Bracken. The result of following the algorithm above was that we obtained expected responses for all observed data falling in severely disadvantaged areas, under Defra’s “environment only” policy scenario.

Results

Fitted models

For the model of bee nectar plant diversity, the final set of chosen covariates comprised: mean annual temperature, mean monthly accumulated rainfall, a first order interaction between mean annual temperature and mean monthly accumulated rainfall, nitrogen deposition, Broad Habitat type, altitude, woody cover and patch geometry. For the model of butterfly nectar plant diversity, the final set of explanatory variables was: mean annual temperature, coincidence within Entry Level Stewardship (ELS) in England, nitrogen deposition, Broad Habitat type, altitude, woody cover and patch geometry. Note: the variable geometry is a proxy for either total length of linear features in each 1km square or total polygon area depending on the Broad Habitat and plot type analysed. Both models also included a spatial trend surface in the form of an interaction term for the spatial location of each 1km square and a random affect of survey square on between plot variability.

Model predictions

Based on the fitted model, predicted numbers of nectar producing plants per $4m^2$ plot shown by each Broad Habitat (Figs 5.10 and 5.11) show the geographic distributions of habitat types as well as the differences in predicted richness between them. Note that these predictions have been made only at the locations where the data was initially recorded and are shown much larger than true size to enable visualisation. As expected Rivers and Streams (bankside vegetation), Boundary and Linear Features, Calcareous Grassland, and Neutral Grassland are the richest habitat types. While it is very apparent that Broad Habitat is the major factor explaining variation in nectar plant diversity, the fact that the two models express geographic variation in nectar plant diversity *within* each Broad Habitat indicates the clear importance of the other covariates in the models (Figs 5.10 and 5.11).

Scenario testing

Combining the scenario testing approach with a model based approach to mapping, we can map the estimated response in Severely Disadvantaged Areas under the policy scenario (Figs 5.12 (a) and 5.12 (b)). Although the scenario required adjustment of the Broad Habitat for each polygon, the mapping resolution is at the 1km square level. Hence mean values for the 1km square may often not change appreciably. Given the finer resolution at which the model can operate, it would be feasible to zoom in on a specific area and map predicted change in potential nectar plant diversity in greater detail. Figs 5.12 (a) and 5.12 (b), simply demonstrate the scope for selecting specific regions of Britain based on Land Cover Map (LCM) habitat composition, applying the model and upscaling the results to produce mapped outputs.

A detailed assessment of the impact of the scenario in each square requires analysis of the distribution of differences between nectar plant diversity at the start and nectar plant diversity after land cover change across the distribution of possible outcomes. The bootstrapping approach used in this example, is a useful way of producing a large database of possible outcomes. It has two advantages; first, because there are many possible permutations of loss and gain among the polygons in a 1km square, some of which may yield a greater or lesser benefit than others, it is useful to generate a large distribution of possible changes. This database of outcomes can then be searched to determine the configuration of habitat transfers in particular locations that led to the greatest or least predicted gains in nectar-plant diversity. The second advantage is that the approach helps quantify the uncertainty among the predicted outcomes.

As a first pass, the dataset of differences between starting and ending diversity were examined to identify particular transfers of habitat that were likely to yield greater potential nectar plant diversity across 1km squares. Results showed that on average, changes that involved loss of Coniferous Woodland could generate the largest potential increases in nectar plant diversity for butterflies whereas for bee nectar plants there was little to choose between donor habitats in terms of the dividend gained from conversion. Again, this analysis was based on mean changes over the total number of simulated transfers between pairs of habitats. The database of predictions therefore offers much greater scope for identifying particular configurations of change that seemed to yield higher gains by examining cases in the upper tails of the distribution of predictions.

5.8 Discussion

Further development and application

The models developed here show great potential for further application. They have the advantage of being trained on high quality, fine grained datasets but recorded from the entirety of the Britain. This means there application at small regional scales is only limited by the availability of covariate data with which to make predictions. Since the GAMM are relatively quick to build, it is also possible that the models can be updated easily should additional covariates become available, for example further information on agri-environment scheme uptake in other parts of Britain. The existing models could however be readily used to address other existing scenarios of change in land use in Britain. Moreover, the inclusion of predictors for land-use, landscape structure and climate offers the intriguing prospect of being able to apply multiple scenarios in the same prediction, for example land-use plus climate change. The flexibility of the models also opens up the possibility of applying more ecologically interesting rules for land cover change in particular areas; for example taking into account adjacency of polygons to beneficiary habitats or accounting for differences in size of donor polygons or differences in patterns of transfer linked to greater detail in the scheme options in any particular area.

Although promising, a number of caveats do apply to further model application. An obvious one is that the model predictions are based on instantaneous responses to changes in predictors. That is, there is no modelled lag effect that accounts for the time taken to reassemble plant communities following land use change. Difficulties in restoring ecological conditions and differences in recruitment probability from local plant species pools are two key issues among others to be considered. Hence, predictions are more accurately termed *potential* nectar plant diversity. Much information exists on the timescales achievable for habitat restoration from different starting points and this information could be easily reviewed and applied to the predictions to add an element of temporal uncertainty to changes in nectar plant diversity expected to arise from different scenarios. It is also feasible that these constraints could be formally built into the models using knowledge on local species pool composition.

The application of this model-based approach to scenario testing is possible due to the extensive research and reporting of the particular policy scenario in question. In order for methods of this kind to be of greater use in the future, the importance of detailed, quantitative scenarios is paramount.

Chapter 5: Online appendices

5.1: Frequency tables of nectar plants in CS vegetation plots.

Figure 5.10: Results of prediction of nectar plant diversity for bees in vegetation plots modelled using data from Countryside Survey in 2007, shown by Broad Habitat type. Plots are 4m^2 for all Broad Habitats except for the Boundary and Linear Features and Rivers & Stream Broad Habitats (1x10m). (Note: predictions are shown much larger than true plot size to aid visualisation).

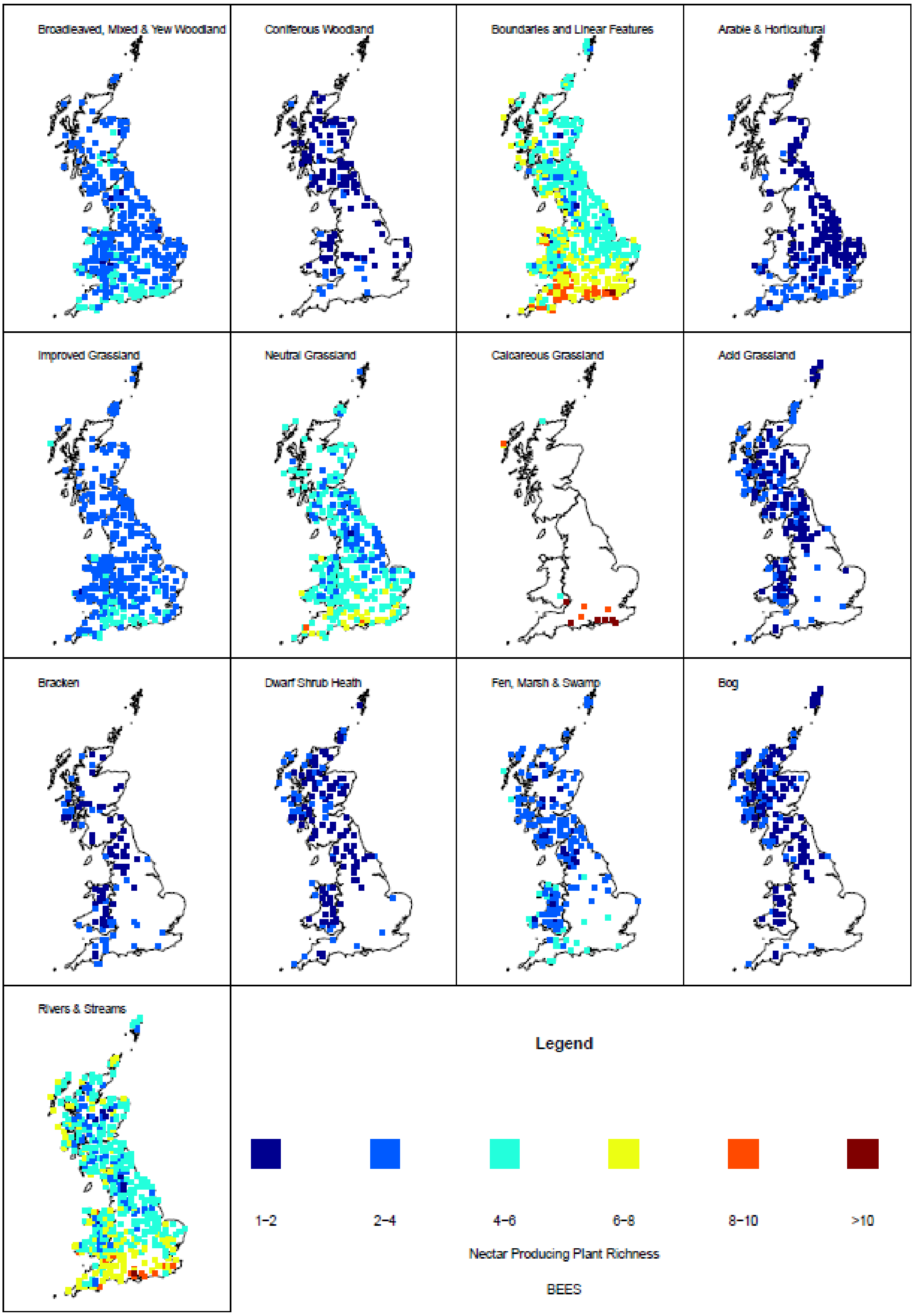


Figure 5.11: Results of prediction of nectar plant diversity for bees in vegetation plots modelled using data from Countryside Survey in 2007, shown by Broad Habitat type. Plots are 4m^2 for all Broad Habitats except for the Boundary and Linear Features and Rivers & Stream Broad Habitats (1x10m). (Note: predictions are shown much larger than true plot size to aid visualisation).

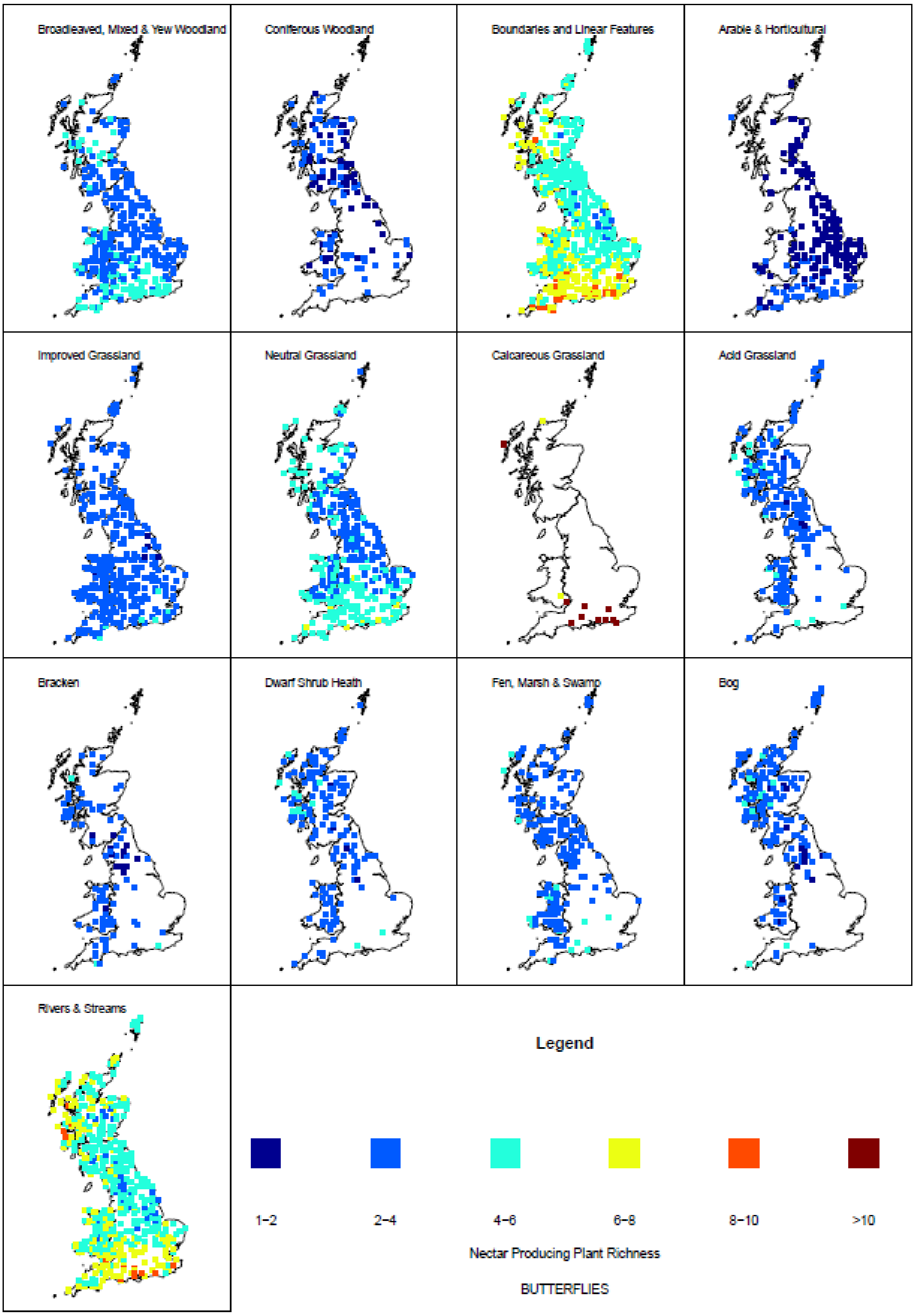
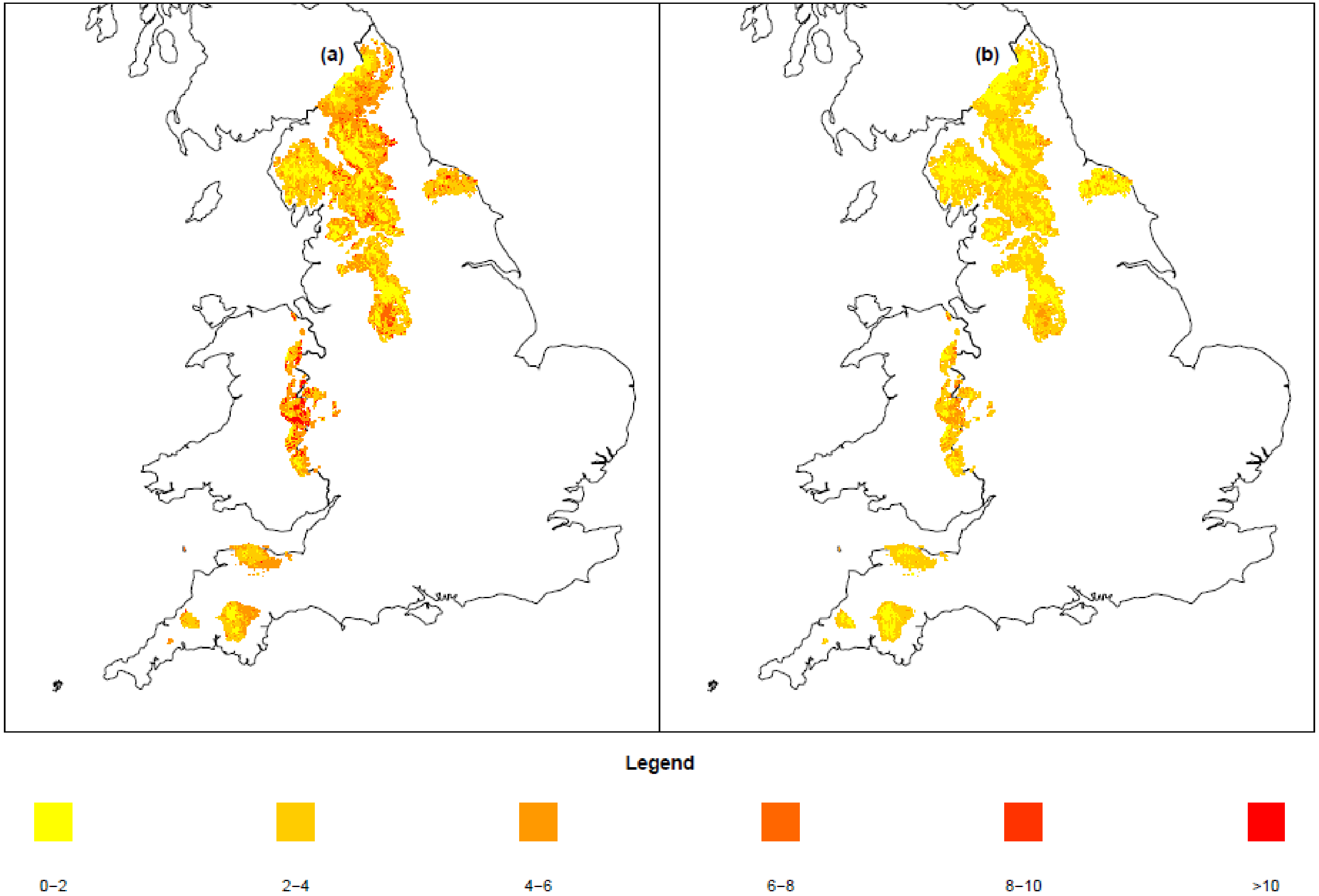


Figure 5.12: Predicted potential nectar plant diversity for bees (a) and butterflies (b) under the Defra "environment only" policy scenario. Predictions were made using Land Cover Map 2000 and climate data across Britain so they can be mapped to all Severely Disadvantaged Areas at a 1km square resolution including those outside of Countryside Survey squares in 2007.



Chapter 6: Using Countryside Survey data to quantify the cultural services of English landscapes

L.R. Norton, H. Inwood, A. Baker

Summary

- This chapter reports on a preliminary exploration of the potential for using quantitative Countryside Survey data (CS) on habitats and landscape features alongside qualitative data to test preliminary spatial expressions of cultural services across England. Relationships between National Character Areas (NCAs)[41] and Countryside Survey landclasses[42] are examined. Maps of measures of landscape complexity and of measures of underpinning ecosystem services are briefly considered.

- Countryside Survey data were integrated with qualitative survey data to provide measures of cultural services. Maps of these services were made by extrapolating CS square data using CS landclasses.

- Maps of variables underpinning ecosystem services indicate strong relationships between different measures of biodiversity.

- Countryside Survey data can provide a flexible dataset for policy makers wishing to understand relationships between landscape variables and cultural services.

- It is possible to extract criteria measured in Countryside Survey which may provide information on key characteristic of National Character Areas at relevant regional levels.

6.1 Introduction

This chapter consists of three preliminary approaches towards an exploration of the potential use of Countryside Survey (CS) data for the measurement of cultural (Box 6. 1) and other ecosystem services. The publication of a recent report by Natural England (NE) on 'Experiencing Landscapes'[43] led to the

[41] http://www.naturalengland.org.uk/ourwork/landscape/englands/character/areas/default.aspx
[42] Merlewood Research and development paper no.115.
[43] http://naturalengland.etraderstores.com/NaturalEnglandShop/NECR024

possibility of exploring how CS data may be used alongside the qualitative information recorded as part of the NE research, to provide quantitative measures of the cultural services offered by English landscapes. Work on linking CS landclasses to National Character Areas (NCAs) was required to attempt this and that work is presented here to demonstrate potential approaches for joint work using these two different landscape classifications. The other area of work covered in this chapter explores the potential for mapping a measure of habitat complexity, which may at some level be considered an aspect of cultural services, but is in itself a measure of underpinning biodiversity. A map of habitat complexity is provided alongside maps representing other aspects of underpinning biodiversity.

Cultural Services ***Box 6.1***

The non-material benefits that people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation and aesthetic experience, including, for example, knowledge systems, social relations and aesthetic values. These may also be seen as 'cultural benefits' as they directly relate to changes in human welfare.

6.2 Biophysical measurement

Two separate composite biophysical measures have been selected. One of these measurements, charismatic landscapes, relates to a recent report commissioned by Natural England (NE) (The Research Box 2009), which reported on an extensive qualitative social research study designed to provide baseline evidence of the cultural services and experiential qualities that landscapes provide to society (more detail in section 6.3). The second measure, 'landscape complexity' has been selected for its potential value in representing habitat complexity at the landscape level but could be viewed as a cultural service, where complex landscapes are preferred.

'Charismatic landscapes'

The term *'Charismatic landscapes'*, is used as a descriptor of ecosystem services to encapsulate several variables describing cultural ecosystem services. This work was carried out jointly with the researchers (The Research Box) who carried out the NE qualitative research on the 'Experiencing Landscapes' study. The work aimed to identify a potential approach to mapping landscape quality as a measure of cultural services, by using the qualitative information collected by Research Box in conjunction with CS data and Ordnance Survey data on elevation. The work focused on: two key habitats, broadleaved woodland and water, the area of sea (as a surrogate for coastal area) and variables describing the elevation of land within a square. All of these variables had featured strongly as important components of valued landscapes in the qualitative research work. The focus of the qualitative research was England only and more specifically the eight National Character Areas in Table 6.1.

The CS dataset contains a large range of data representing the many different forms that habitat components take. For example, woodland may be represented by a small clump of trees or by a larger area of habitat, it may be broadleaved or coniferous. Water in the landscape comes in many forms and may, for example, be dependent on the relief of the landscape with ditches and ponds typical of some areas and waterfalls and lakes typical of others.

The Research Box were provided with a range of potential measures from the CS database for the chosen habitats and variables and finally selected the following set of variables:

1) *Woodlands: Area of (Broad Habitat (BH)) Broadleaved and Mixed Yew woodland (km^2).*
2) *Water: Area of (BH) Standing Open Water and Canals, Area of (BH) Rivers and Streams (km^2).*
3) *Relief: Mean altitude (m) and Relief (m) (difference between maximum and minimum altitude).*
4) *Coastal area: Area of (BH) sea (km^2).*

All areas were expressed as a proportion of the landcover of the 289 1km x1km sample grid squares in the CS dataset for England. Using these data The Research Box derived further variables and eventually scores as follows.

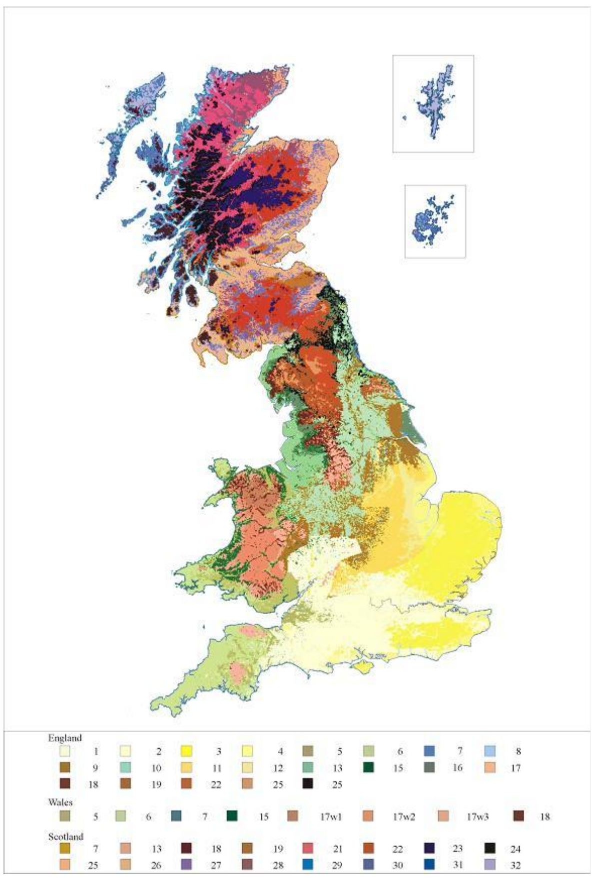
An initial issue was the degree of variability within the CS data. For example, within landclass 1e (undulating country, varied agriculture, mainly grassland) the extent of woodland within the sample km squares ranged from 0.7% to 48.9% – the latter could be considered as woodland, the former probably not. In view of this variability, it was decided to calculate a ‘probability’ that each landclass contained the features in question. This probability informed the later judgement that was made on the degree of cultural service associated with the landclass.

For each landclass, a judgement was made by the research team about the extent to which cultural services are delivered. This judgement was made separately for each of the CS data variables listed above, using the following scale: none (0), low (1), medium (2), high (3). For most areal variables the score was higher, the greater the probability of finding the Broad Habitat (BH) in question within the landclass and the greater the extent of that BH (in % cover). So for water BHs and for sea the more likely you were to find it in a landclass and the greater the extent of it within that landclass the higher the score. The exception was Broadleaved Woodland, where the research indicated that extensive cover was less highly valued than a mixture of woodland and open areas (e.g. fields).

The scores for each of the variables within each landclass were summed to provide an overall score (or cultural service measure) for that landclass. NB landclass divisions are at least partly dependent on altitude as an underlying variable (see Box 6.3).

National Character Areas	**Box 6.2**

England has been divided into 159 National Character Areas[1] (formerly Joint Character Areas) which provide a national spatial network used for a range of applications including the targeting of Natural England's Environmental Stewardship Scheme. Character descriptions of the NCAs are published in eight regional volumes highlighting the influences which determine the character of each area, e.g. land cover, buildings and settlement.



Landclasses **Box 6.3**

Landclasses are the landscape categories underlying the sampling stratification for Countryside Survey. They are derived from a statistical analysis of 40 environmental variables including climate, soils, topography and geology which groups like 1km squares across GB into distinct groups (land-classes). England was divided into 29 land-classes in 2007 (Fig. 6.1) derived from an original set of 32 landclasses used to describe the whole of GB in the first Countryside Survey in 1978 (Bunce *et al*.1996).

Figure 6.1: Countryside Survey land-classes for England, Wales and Scotland in CS in 2007.

Additional work - Matching National Character Areas with Countryside Survey landclasses

In addition to work on landscape quality a desk-based examination of the relationships between National Character Areas (previously known as Joint Character Areas, see Box 6.2) and Countryside Survey landclasses[44] (see Box 6.3, Fig. 6.1) was carried out. This work helps to contextualise the National Character Areas in terms of land classes.

The integrated assessment database referred to in Chapter 1 incorporates a range of datasets alongside the Countryside Survey data. These include area datasets downloaded from the Magic website[45]. The “EnglandCharArea” dataset contains the relevant spatial data delineating the NCAs. Spatial analysis was performed using ESRI ArcGIS 9.2 (ESRI, 2006). In order to examine the land classes in the context of NCAs, the 1km resolution raster of land classes was converted to a vector format and intersected with the NCA polygons for England. The attribute table for the intersected features was exported to a SAS format to examine the distribution of landclasses within NCAs.

Landscape complexity

Landscape complexity and its potential role in the provision of ecosystem services is of interest. It also has potential as an indicator of cultural services, as shown by the Natural England qualitative research which showed that varied landscapes are valued over uniform ones (although the research also found that ‘simplicity’ can be highly valued in some landscapes). CS data are able to reflect landscape complexity because of the vast range of measures recorded at each 1km square.

In an attempt to map the potentially important aspects of complexity/habitat diversity in a 1km square the following measures were extracted and combined to provide a possible index of landscape complexity:

- Numbers of polygons per square * Number of different vegetation types recorded per square (this is below Broad Habitat level, at the primary attribute level, e.g. within the Broad Habitat Acid Grassland, there are two possible primary attributes – Moorland grass or Acid grass).
- Length of linear features per square * Numbers of different types of linear feature per square (including; streams, hedges, lines of trees, fences, walls etc).
- Numbers of point features per square * different types of point feature per square (including; individual trees, ponds, patches of scrub, small buildings etc).

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[44] Merlewood Research and development paper no.115 The ITE Land Classes
[45] http://www.magic.gov.uk

Two other alternative measures, which represent biodiversity at the species rather than the habitat level, were derived from the CS data for comparison with the above index of habitat complexity. These measures were:

- Total species richness per 1km square (the total number of species recorded across all plots within a square).
- Mean species richness per plot for each 1km square.

To map habitat and species diversity at the GB scale, the vectorised landclass features were used as a base map. Diversity means for each landclass were determined by calculating the mean for each of the habitat and species diversity variables of the survey squares in that landclass. These mean values were joined to the landclass features and maps produced by colour coding the landclass areas based on their mean values.

6.3 Rationale for selection

The composite biophysical measure chosen to represent charismatic landscapes directly reflects the 'Experiencing Landscapes' qualitative research carried out on cultural services by The Research Box for Natural England[3]. The work described here was specifically targeted at investigating how CS data could be used to quantify cultural services as measured in this qualitative research. (The qualitative 'Experiencing Landscapes' research was aimed at updating England Natural Character Area (NCA) descriptions and supporting the implementation of Natural England's All Landscapes Matter policy).

Brief description of relevant information from the 'Experiencing Landscapes'[3] report

The study researched more than 150 members of the public living in, working in or using eight selected Natural Character Areas across England (Table 6.1). Eight cultural services adapted from the Millennium Ecosystem Assessment[46] were examined through the research. Of the twenty different features focused on in the study 'water' and 'woodland' were two that greatly enhanced people's experience of landscape. Water was valued for completing the beauty or tranquillity of a place, providing peace and symmetry and, contrastingly, for being exhilarating and for its sounds. Woodlands were considered as treasured habitats which were characteristically British as well as relaxing, magical,

[46] http://www.millenniumassessment.org/en/Global.aspx Sense of History (or heritage), Sense of Place (identity, home), Inspiration (stimulus), Calm (relaxation, tranquillity), Leisure and Activities (recreation), Spiritual, Learning (education), Escapism (getting away from it all).

attractive to look at (particularly broadleaved), comforting and sometimes frightening. Another key variable was the ‘relief’ of the landscape with ‘hilly’ landscapes providing views and a feeling of ‘energy’.

People also felt that it was important to be able to see patterns, different colours, layers and views of boundaries, preferring complex, well-punctuated landscapes to simple ones.

The composite biophysical measure chosen to represent landscape complexity has been included in this chapter in part because of its potential importance as a measure of cultural services, although it is recognised that it is only one measure of a very complex aspect of landscape value. The measure has also been included in this chapter because of its ecological interest and because early investigations have revealed some interesting patterns which merit further exploration. Habitat diversity is an aspect of biodiversity and there are interesting questions about the relationship between measures of habitat and species diversity which are touched upon in this work.

Table 6.1: The National Character Areas used for research on ‘Experiencing landscapes’.

NCA no	NCA description
9	Eden Valley
15	Durham Magnesium Limestone Plateau
27	Yorkshire Wolds
42	Lincolnshire Coast and Marshes
111	North Thames Basin
119	North Downs
145	Exmoor
148	Devon Redlands

6.4 How are landscape measures linked to cultural ecosystem services?

As referred to above, this work is closely linked to qualitative work undertaken for Natural England on ‘Experiencing landscapes’[3]. This work specifically addresses the role of landscape in the provision of cultural services.

Cultural services are defined as the nonmaterial benefits that people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation and aesthetic experience, including, for example, knowledge systems, social relations, and aesthetic values (see Box 6.1). It is widely recognized that landscapes provide people with these services but the extent to which particular landscape types or features differ in the extent of their provision are difficult to measure. An ecosystem service cascade (Haines-Young and Potschin 2007) for landscape quality indicates how biophysical measures may be linked to functions and services and their value (Fig. 6.2). It should be noted that whilst there are now numerous measures for the majority of ecosystem services, there are very few for cultural services (Feld *et al.* 2009).

Landscape complexity as a measure of habitat diversity may be viewed (like biodiversity) as underpinning all other ecosystem services.

6.5 Current status across GB

Mapping 'charismatic landscapes'

A map showing the scores calculated for each landclass in England is shown in Figure 6.3[47]. The map also includes the boundaries of the 8 National Character Areas (NCAs) in which the qualitative work took place.

In simply summing the scores, the research team made an assumption that all variables are equally important, although there is no definitive research that would support this within the current context.

Further issues to note are:

- No distinction was made between the eight separate cultural services examined in the Natural England research – the judgement 'score' derived for each 'feature' within each landclass relates to all in combination
- Scores relate to local conditions, i.e. those that exist within the km square. Many cultural services relate to the landscape features that can be seen – not necessarily in the immediate vicinity (and sometimes at a considerable distance). This mapping takes no account of features that are visible but which do exist within the grid square

[47] Urban areas have been excluded from the map. CS does not survey 'Urban' areas – 1km squares with greater than 75% urban cover. Additionally, the Experiencing Landscapes study did not cover urban areas in the qualitative social research undertaken.

- Not all landscape features that deliver cultural services were taken into account in this mapping exercise (including historical features; hedges, walls and lanes; field systems; villages).

A feature of the map shown in Figure 6.3 is that all scores are contained with a narrow band and, hence, illustrate that cultural services are widely spread across England. This supports a key finding from the Experiencing Landscapes research that all landscapes (even those that are local and mundane) are important for delivering cultural services. However, it would appear that some 'pinnacle' landscapes (e.g. the Lake District and the South Downs) seem not to be scored as high as one might expect – probably as a result of the points raised above.

The map has been tested against what is known about the extent of cultural services delivery within the eight pilot NCAs covered in the 'Explaining Landscapes' research – broadly speaking, the scores derived for each NCA, based on the landclass incidence within the NCAs, is supported by the research findings.

It should be stressed that the map of cultural service scores in England is an initial attempt at examining the spatial distribution of cultural services. Further work, to address some of the known weaknesses of the current (and the issues identified above) would be recommended.

Matching National Character Areas with Countryside Survey landclasses

There are a total of 21 landclasses in England and 159 National Character Areas covering approximately 128,380 square km. The number of landclasses in each NCA was on average 6.7 but varied between 0 and 16. The number of NCA's represented in each land class was on average 11 but varied between 4 and 26.

A detailed breakdown of landclasses (together with a brief description) by NCA is provided in Appendix 6.1. Table 6.2 shows a detailed breakdown of the comparison between NCA 7 and its component landclasses and isolates the Countryside Survey measures which may be used to assess landscape change in NCA's.

Mapping landscape complexity

The results of the mapping exercise for a) Habitat diversity, as a measure of landscape complexity, is presented alongside species level biodiversity measures; b) Total species richness per 1km square (the total number of species recorded across all plots within a square) and c) Mean species richness per plot for each 1km square across GB, in Figure 6.4.

Table 6.2: Landclass composition of National Character Area 7 (West Cumbria coastal plain) and appropriate Countryside Survey measures.

Landclass	***% of NCA***	***Landclass description***	***NCA description***	***CS measures***
7e	4	Coastal with variable morphology and vegetation.	• Varied open coastline of mudflats, shingle and pebble beaches with localised sections of dunes, sandy beaches and sandstone cliffs. • Lowland river valleys with limited semi-natural ancient woodland, lowland raised mires and expansive estuarine landscapes with a range of intertidal habitats. • Gently undulating or flat improved pasture with hedgerows, wind-sheared trees and wire fences, occasional woodlands and copses, wetlands and herb-rich meadows. • Open agricultural landscapes that have extensive views to the higher fells in the east. • Extensive urban fringe areas within the coastal belt.	Broad and Priority Habitat Extent and condition (although not for inter-tidal habitats) Hedgerow length and condition. Numbers, species and age of individual trees.
8e	16	Coastal, often estuarine, mainly pasture, otherwise built-up.		
15e	4	Valley bottoms with mixed agriculture, predominantly pastoral.		
13e	54	Somewhat variable land forms, mainly flat, heterogeneous land use.		
16e	19	Undulating lowlands, variable agriculture and native vegetation.		
18e	1	Rounded hills, some steep slopes, varied moorlands.		
5e	1	Lowland, somewhat enclosed land, varied agriculture and vegetation		
6e	1	Gently rolling enclosed country, mainly fertile pastures.		

6.6 Discussion

It is clear that there are many possibilities and potential approaches that could be adopted to provide spatial expressions of cultural services across Great Britain. The approaches here should be considered as exploratory rather than providing any absolute answers.

'Charismatic landscapes'

This exercise indicates that the CS data is able to respond flexibly to independent measures of cultural services. There is clear potential for using CS to look at cultural services in a quantitative way if evidence is available to provide focus on the specific landscape features which render those cultural services. The issue of scale is important. The scores have been extrapolated across England, but the qualitative work was focused at the local (NCA) level, potentially indicating that the measures are only relevant at the NCA scale. It may be argued that landscape preferences tend to be broadly relevant for universal features, such as water, although conversely some features may only be experienced locally, e.g. ditches in lowland Britain, or high fells in Scotland. In the example mapped here, ditches were excluded from the 'water' score because they were not seen to be important in delivering cultural services amongst the people researched. It is apparent that scale needs to be explicitly addressed in any exercise of this kind.

The other key issue is how to convert a range of biophysical measures (with potentially diverse units of measurements) into a score representing a cultural service. Initial attempts used raw data to avoid any subjectivity in the relative importance of different aspects of the woodland and water variables on which the study was focused. However, this led to relatively high 'weightings' for some features, in particular lengths of streams and ditches as compared to areas of Broad Habitat. It should be stressed that cultural services are subjective – they are perceptions of the landscape, as interpreted by people – and so are probably not ideal subjects for a wholly quantitative approach. The relationship between the extent of features (woodland or water, for example) and the 'amount' of cultural service delivered is not straightforward and any scoring based wholly upon a measure of the existence of biophysical features is unlikely to be accurate.

Ideally, biophysical measurements would closely reflect the features focused on in any qualitative research carried out in this area. In the work described here, this was only partially the case, since the qualitative research objectives lay elsewhere. Despite this, there are some pertinent findings that show the complexities involved. For example, when 'water' is present in a landscape, it is valued in the form of streams, a waterfall, or a lake – but not in roadside ditches, or farm ponds.

It is clear that CS data can provide appropriate measures for many of the cultural services which the landscape provides, including some features that provide a 'sense of history' or identity, spiritual benefits and inspiration to those providing places for escapism, relaxation, education and recreational activities. However, not all of the landscape features found to be important are in the data, e.g. aspects of the built environment[48], so, whilst the CS data provides a good starting point, on its own it is unlikely to be sufficient to develop a comprehensive map of cultural services. One important issue is the limitation of the CS data in relation to distant views which the Experiencing Landscapes research has shown to be important. If this work was to be taken further it may be possible to use the Land Cover Map to address this issue. Similar research in the Netherlands also revealed the importance of views when investigating the potential use of landscape data for measuring landscape quality (Farjon *et al.* 2009). The work also drew attention to the importance of social science work (questionnaires) to accompany landscape measures in the valuation of landscapes.

Matching National Character Areas with Countryside Survey landclasses

Fundamentally CS landclasses and Natural England's NCAs were developed for different purposes although they both describe landscape. Landclasses are based on underlying environmental variables so that areas of the same landclass can be widely separated from one another. For example, landclass 8 which is described as 'coastal, often estuarine, mainly pasture, otherwise built-up' is found in East Anglia, southern England and on north-western and south-western English coasts[49]. In contrast, a key feature of NCAs is that they are contiguous and regionally distinctive. CS was originally designed to explore landscape patterns and change at a national scale, but has over recent surveys been adapted to enable country level reporting for Scotland, England and Wales. NCAs are designed to provide a basis for maintaining and improving landscape character on a regional scale. This basic inconsistency in design inevitably impacts on the extent to which landclasses can be matched to NCAs.

Previous work focusing on the potential use of Countryside Survey information for measuring landscape change within Joint Character Areas (the pre-cursors of NCAs) recognised the issues which arise because CS is designed as a national survey[50]. The study concluded that for some purposes, the use of CS data was limited, because it is "essentially national, sample-based data which cannot be used to make statistically robust estimates at fine geographical scales". However, preliminary work in the study which used associated Character Area Descriptions

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[48] Sense of Place and Social Capital and the Historic Built Environment (2009) Report of research for English Heritage, available at http://www.english-heritage.org.uk

[49] Merlewood Research and development paper no.115 The ITE Land Classes

[50] Countryside Quality Counts - Tracking Change in the English Countryside. Constructing an Indicator of Change in Countryside Quality 2004– Roy Haines-Young, Julie Martin, Dominic Tantram, Carys Swanwick. http://countryside-quality-counts.org.uk/publications/CQC-1990-1998-FinalRep.pdf

to identify the types of change that are 'consistent' or 'inconsistent' with the general character of the area appeared to produce figures which were broadly consistent with those derived from alternative data sources. The work indicated that patterns of habitat change at the square level could provide information relevant for the understanding of patterns in NCAs.

More work is required to explore the potential for using quantitative data at a landclass level to describe the landscape qualities of the NCAs. Both landclasses and NCAs are fundamentally tools for stratifying landscape based on environmental variables, albeit at different scales. Inevitably there is common ground between the two classifications. It is, however, important to recognise the limits to sensible use of CS data given the survey design. Using CS squares to provide data for NCAs would be inappropriate for NCAs with limited numbers of sample squares. A more effective way of using CS data to report on NCA's might be to extract criteria measured in CS from the NCA key characteristics for each character area (see Table 6.2) and use that data at a relevant regional level to assess whether changes in CS measures indicate change consistent with maintaining NCAs. The 'relevant regional level' is likely to refer to a level for which the sample size (of CS squares) is adequate to detect statistically significant change and may comprise, for example, an aggregation of NCAs. A number of regional masks including the National Character Areas are provided on the CS website[51] for which data are available for download.

Landscape complexity

This exercise was an attempt to look at habitat complexity at a landscape level across Great Britain as a measure underpinning all ecosystem services as well as a potential indicator of cultural services in a landscape. Species richness is also investigated alongside habitat complexity with both representing potential measures of biodiversity. CS uniquely enables large scale comparisons between landscape and vegetation measures across GB. Ecological complexity is recognised as important in making ecosystems more resilient to potential drivers of change. In managed landscapes like England, policy incentives (e.g. Entry Level Stewardship options)[52] to increase the complexity of otherwise uniform landscapes reflect the importance of maintaining habitat diversity. Recent work by Lovell *et al.* (2009) includes a review which reveals important opportunities to improve the quality of the landscape matrix by increasing spatial heterogeneity through the addition of seminatural landscape elements designed to provide multiple ecosystem services in both urban and agricultural settings.

Mountainous regions across GB exhibit low habitat complexity, whilst areas of Cornwall, Wales and the Welsh border country are constituted of complex landscapes (Fig. 6.4). The lack of habitat diversity in mountainous regions should not be taken as a negative trait as they tend to contain relatively few habitats and no linear or point features notable in CS. In order to assess whether habitat

[51] www.countrysidesurvey.ac.uk
[52] http://naturalengland.etraderstores.com/NaturalEnglandShop/NE226

complexity is appropriate to particular habitats/regions it would be necessary to explore potential approaches such as the Common Standards Monitoring species approach for appropriate diversity which has been used in Chapter 4, or use, for example, guidance on habitat management for BAP species[53].

If using habitat complexity as a measure of cultural services, it would be advisable to explore weightings to balance the emphasis placed on the different aspects of complexity (as for charismatic landscapes), alongside a more detailed understanding of the aspects of complexity which yield cultural services (e.g. do individual trees in hedges provide more value than in-field trees). Additionally, there is scope for calculating the score using different approaches and for normalising the data. These possibilities will be explored in future work in this area and outcomes will depend heavily on whether the objectives being pursued relate to cultural or other ecosystem services.

Species richness is an ecological measure which is not generally valued by the wider public, unless it is related to 'typicality' of a particular habitat type, e.g. species rich flower meadows. In contrast, some ecologists use species richness as a measure of habitat quality when the species present are appropriate to the habitat in question. Species richness at a landscape scale is likely to represent heterogeneity or complexity of landscape structure and/or the presence of high quality habitats. It is also associated with geographical position, with the south of Great Britain containing greater numbers of species due to its warmer climate and its proximity to mainland Europe.

As well as exhibiting low habitat complexity, mountainous regions across GB also contain low numbers of species at the 1km square scale (Fig.6.4b). Upland landscapes are generally species poor because of acid soils and a cooler climate which limit the range of species/communities that can survive there. Patterns across Great Britain as a whole, show strong similarities between species richness (Fig. 6.4b) and habitat complexity (Fig. 6.4a) but require closer investigation to identify inconsistencies.

At the plot level, species richness is negatively related to habitat diversity, with higher diversity relating to lower numbers of species (Fig. 6.4c). This may, in part, be due to the intensity of management. Intensively managed areas can have more habitats than less intensively managed areas, although both the most intensively managed landscapes (e.g. arable farmland in Cambridgeshire) and the least managed (e.g. the Highlands of Scotland) have low habitat diversity. High habitat diversity in the lowlands may be associated with many small patches of habitat compared to the large blocks of habitat in the uplands. These results are preliminary but combining spatial and plot data from the CS survey has revealed some interesting patterns.

[53] http://naturalengland.etraderstores.com/NaturalEnglandShop/NERR024

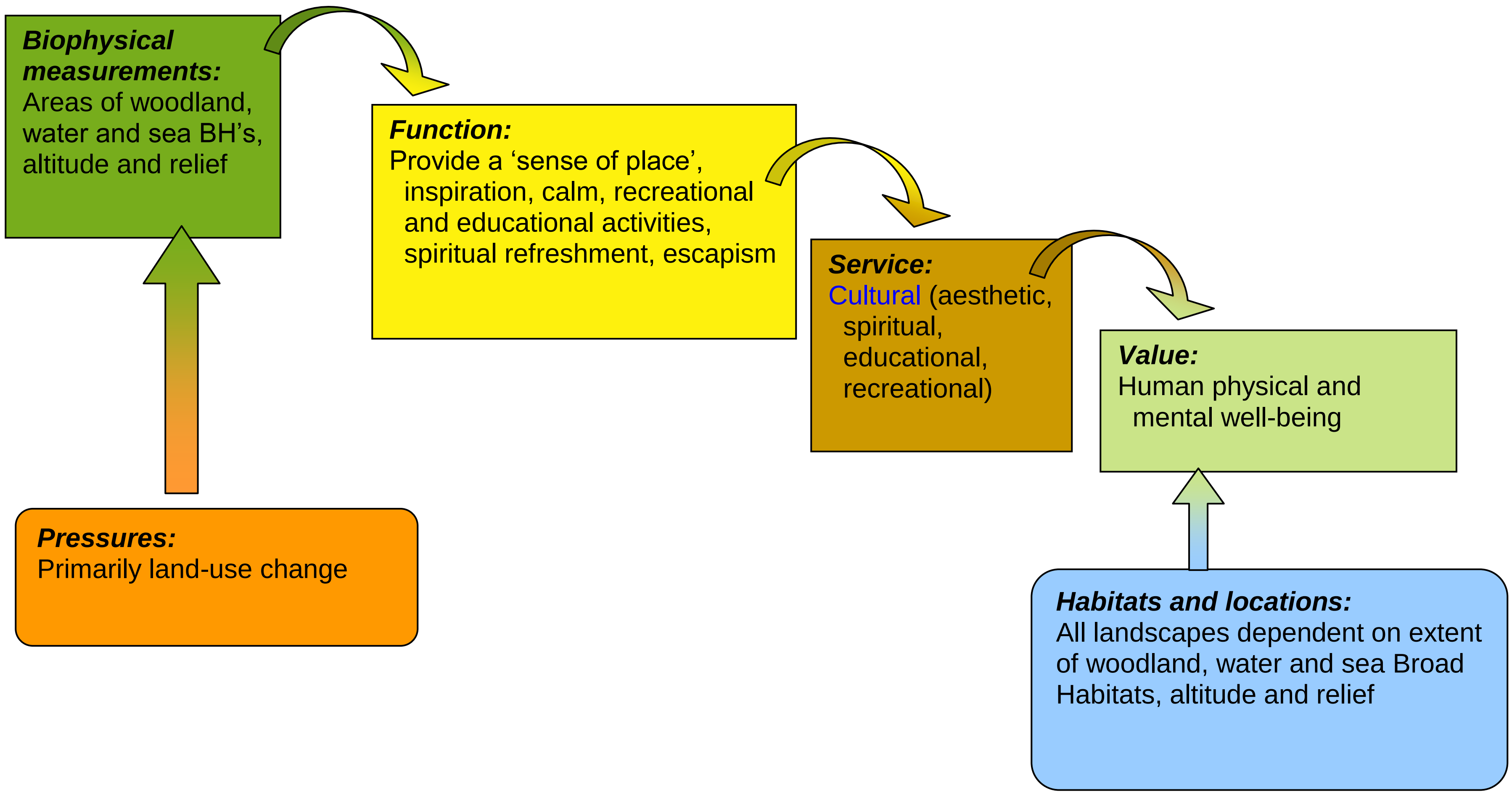
6.7 Potential areas for further work

- Further work alongside social scientists to provide/use relevant quantitative measures of the cultural services offered by the landscape using CS data potentially alongside Land Cover Map and other national datasets.
- Refining habitat diversity measures.
- Identifying ‘appropriate habitat diversity’ measures.
- Exploring relationships between habitat diversity and appropriate species diversity.
- Exploring the aspects of habitat diversity that generate cultural services.
- Exploring species diversity as a cultural service, for example, is there an ‘appropriate’ species composition for providing certain cultural services (e.g. Oak or Beech/Bluebell woodland).
- Determining the relationships between habitat diversity and species richness at different scales.

Chapter 6: Appendices

6.1: Area contribution of ITE Land Classes to National Character Areas

Figure 6.2: The ecosystem service cascade for 'charismatic landscapes' (after Haines-Young and Potschin 2007).



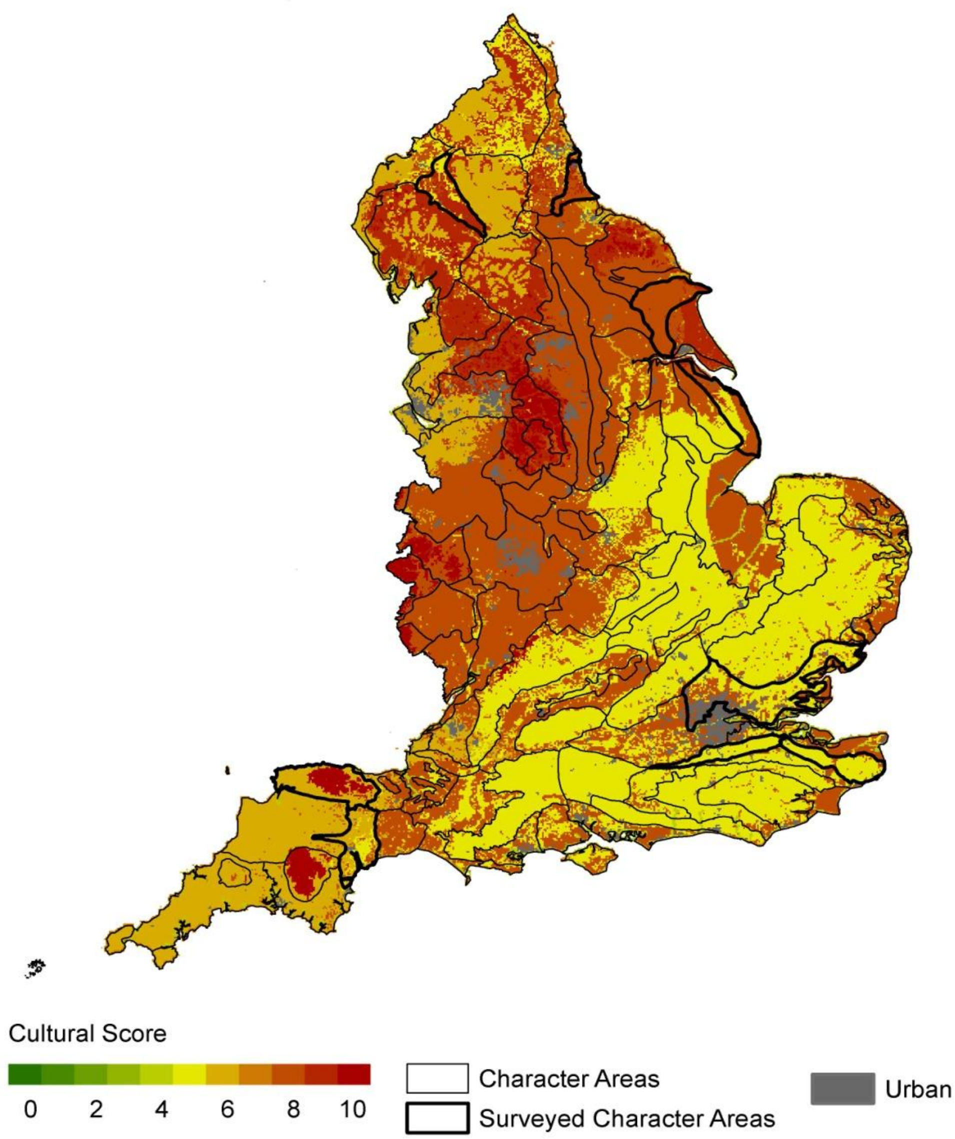


Figure 6.3: Cultural service 'scores' for 'charismatic landscapes' (relating to woodland, water, altitude and coast) for Countryside Survey landclasses (England only). High scores indicates delivery of a greater cultural ecosystem service.

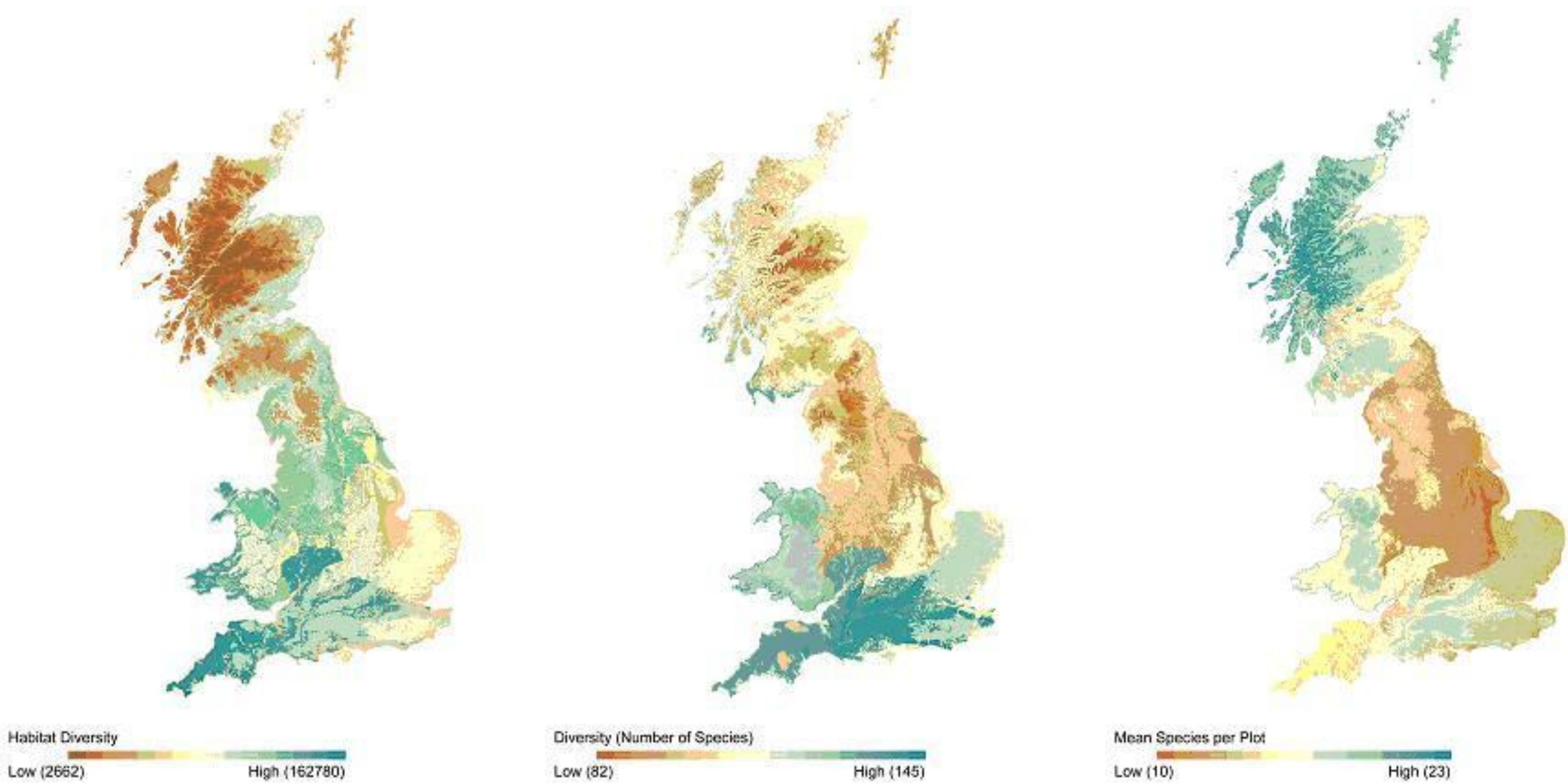


Figure 6.4: a) Habitat complexity/diversity, b) Total species richness per 1km square (the total number of species recorded across all plots within a square) and c) Mean species richness per plot for each 1km square across Great Britain.

Chapter 7: Exploring interactions between ecosystem services

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Summary

- Countryside Survey (CS) data is uniquely able to demonstrate correlative relationships between ecosystem service indicators across Britain. In this chapter CS data is used to explore interactions between these indicators; both synergies and trade-offs.

- Multi-variate analysis of ecosystem services is a useful tool for quantifying patterns of joint spatial or temporal correlation between a mixture of indictaors. By creating an ordination space of ecosystem service indocator values within CS plots across British ecosystems, it is possible to infer the major ecological gradients that appear to constrain biodiversity and its relationships with ecosystem services at the large scale. The approach therefore helps set ecosystem services in context within a landscape, habitat type or country.

- The 1km squares with the highest plant species diversity also had the highest levels of within-1km square variation in other services such as soil carbon. This suggests that high habitat heterogeneity i.e. diversity of both habitats and species will maximise services.

- However, the principal gradient was related to primary production with the most productive plant species and ecosystems at one end and highest values of topsoil carbon within the least productive ecosystems at the other. Each end of this axis was associated with lower plant species diversity.

- Service relationships change according to the scale at which they are observed. For instance, relationships at the GB scale differ from the devolved countries, this is due to variation in ecosystem properties and habitats and variation in policies and land management. Service interactions also vary by habitat according to biotic and abiotic processes.

- Use of a service (demand) also varies spatially. This chapter shows that service supply is concentrated in the north and west of the country whilst population density is highest in the south and east. This creates issues, in some cases the service can be transported (e.g. water supply for big cities like Manchester sourced from uplands and national parks like the Lake District) in others (e.g. recreation) it is more appropriate to try to increase service provision locally.

- In intensively managed landscapes there will be trade-offs between productivity and all other services. This could be compensated for by creating a multi-functional landscape (e.g. proposed options in agri-environment schemes). Short-term losses of productivity may be offset by increases in provision of a wide range of services, long-term sustainability (e.g. by maintaining better soil quality) or by reducing cost of inputs (e.g. fertilisers).

- Measures of diversity representing different ecosystem compartments (i.e. soils, water and vegetation) are mostly positively correlated. There appear to be synergies between variables influencing diversity patterns and policy directed towards high species diversity of one component should benefit other types of diversity. (N.B. total richness of plant species may not be a positive indicator of favourable condition in habitats which have naturally low diversity).

- Relationships between productivity (Specific Leaf Area) and diversity depend on habitat type (and associated ecosystem properties). Relationships are positive in low nutrient community types e.g. Acid Grassland, Dwarf Shrub Heath and negative in communities where productivity is already likely to be high i.e. Arable and Horticulture, Improved and Neutral Grassland. Although this result supports existing understanding of community processes it has not previously been demonstrated using quantitative data in multiple habitats at such a large scale (i.e. GB).

- There was a significant negative relationship between productivity (SLA) and topsoil carbon (LOI). This is a very interesting result and has important implications. It might be possible to estimate topsoil carbon from SLA but does not account for situations where soil and vegetation are in disequilibrium, for example, where there has been recent land-use change. The relationship may point the way to optimising carbon sequestration by balancing vegetation productivity with soil carbon in selected habitats e.g. Neutral Grassland.

- Relationships between topsoil carbon (LOI) and diversity are complex. At a GB scale relationships are positive between topsoil carbon and butterfly nectar plants and negative with bee nectar plants. This seems to reflect the more nitrophilous affinities of the most common bee nectar plants. Patterns in the results from individual countries reflect the composition of habitats within those countries, with for example a higher proportion of moorland and bog habitats in Wales and Scotland than England. Within low diversity habitats topsoil carbon (LOI) is related negatively to total plant diversity (Bog, Fen, Marsh, Swamp and Acid Grassland) where high plant diversity can be a measure of poor condition, although no relationship with Common Standard Monitoring (CSM) Indicators is apparent. It is necessary to move beyond generic statements about relationships between diversity and other services and develop a basic theoretical framework to understand these complexities.

- There are both positive and negative temporal relationships between changes in plant diversity and changes in topsoil carbon but these are only seen in habitats where topsoil carbon stock is low, probably because it is easier to detect a signal in these habitats.

- Higher freshwater ecological quality is associated with high topsoil carbon, low productivity and high biodiversity. Unlike some of the other services freshwater quality was measured at the scale of 1km square so results reflect landscape scale patterns. A trade-off is suggested between productivity and water quality and synergies between freshwater quality, topsoil carbon and biodiversity.

- Protection for conservation purposes (Site of Special Scientific Interest designation) showed an association with high carbon habitats and soil invertebrate diversity. It was less obvious that there was any correlation with plant diversity, possibly because of the high proportion of low diversity upland habitats in the series. The results suggest the possibility that management for conservation might have beneficial effects on provision of other less obvious services.

- With modest further development work, the ordination models could be used to predict the values of ecosystem service indicators for all British 1km squares based on strong multivariate relationships between indicator values and the proportion of Broad Habitats within each square. This could then be extended beyond CS squares to all 1km squares in Britain using the composition of Broad Habitats taken from the new Land Cover Map for 2007.

7.1 Introduction

This report has until now focused on individual services. Attribution of the explanatory factors impacting on each service has involved integration between observed changes in measurements from ecosystem compartments; (i.e. soils, water and vegetation) with a link to service identified and correlated variables driving change. This chapter seeks to bring together these responses in biophysical variables to determine patterns of spatial and temporal intercorrelation between ecosystem services, including both synergies and trade-offs.

The strength of the ecosystem service concept is that it brings together multiple landscape elements (from nature and human well-being e.g. safe drinking water, food, fuel, flood control, aesthetic and cultural benefits) and demonstrates that a balance between these factors is necessary (Armsworth *et al.* 2007; Daily *et al.* 2009). Often the term ‘trade-offs’ is applied to consideration of service interactions. The implication being that you cannot have it all and some services will reduce when others increase. There are also synergies when services interact mutualistically or are enhanced simultaneously by external drivers. Research to understand relationships between services is still at an early stage and development of management strategies to maximise services by managing them as bundles has barely begun.

A number of authors have addressed the need to understand service relationships particularly focusing on links between biodiversity and all of the other services (Chan *et al.* 2006; Naidoo *et al.* 2008; Anderson *et al.* 2009). Conservation policy has focused on protection of areas of high species diversity and it is necessary to determine whether these coincide with high levels of other ecosystem services or whether additional policy strategies/planning for protection of ecosystem services is necessary. Naidoo *et al.* (2008) carried out a global analysis using four services and found that conservation priorities aimed solely at biodiversity do not conserve optimal levels of all services. However, Goldman *et al.* (2008), in a slightly different type of study classified projects into those focused on an ecosystem services approach and those oriented around biodiversity. They found that ecosystem service projects, attracted more funding, included people and landscapes in the studies and were no less likely than the biodiversity projects to include or create protected areas. Chan *et al.* (2006) contradicts this, by suggesting that although targeting ecosystem services can meet multiple service goals and biodiversity efficiently it is not as beneficial for biodiversity as a diversity-centred approach. Anderson *et al.* (2009) in a national study of GB (using a small number of services) found that biodiversity and ecosystem services were not spatially congruent. The diversity of species of conservation concern being in the south and east and other services such as carbon storage in the north and west. However, it does depend on which type of biodiversity is used as a measure, there are SSSI's which protect representative Common Standards Monitoring species of a habitat even if total diversity is not high or species are not necessarily rare or endangered.

So whether biodiversity coincides with high levels of other ecosystem services and whether strategies focused on increasing diversity can also be beneficial for other services is unclear or at least it may not be possible to generalise. It is also interesting to determine whether different components of biodiversity e.g. terrestrial plants, freshwater invertebrates and soil invertebrates are positively correlated. Do specific types of diversity relate to all services in the same way (e.g. soil invertebrates) and can areas be managed to maximise all diversity or is it necessary to prioritise? The relationships between diversity and a range of other services were investigated using CS data; this also involved incorporating membership of an agri-environment scheme or designation for conservation purposes (SSSI) into the analysis.

As discussed in the individual chapters, links between ecosystem services and measurable ecosystem properties can be difficult to quantify. Understanding relationships between services depends on understanding how ecosystems function and how biodiversity relates to ecosystem functioning. In some cases a few key dominant species will drive the functioning of a community and the wider diversity of species and traits is only required when something goes wrong; a perturbation, change in abiotic factors or human intervention causing species loss and driving filtering of the species pool for selection of a different suite of dominants and subordinates (Smith & Knapp 2003; Grime 1998). Although CS has some potential to test complex hypotheses to investigate ecosystem function and diversity further, this potential has not been fully exploited within the constraints of this project. This chapter does, however, incorporate a new metric, that of cover-weighted Specific Leaf Area (SLA). SLA has been shown to correlate significantly (0.78) with Specific Above-ground Net Primary Productivity (SANPP) (Garnier *et al.* 2004) and so it will be used as an indicator of this supporting ecosystem service.

Relationships between services also differ according to the scale of study. Naidoo *et al.* (2008) addressed the issues of scale to some extent; their overall analysis used global ‘ecoregions’ and they also looked within an ecoregion. Despite an ecoregion showing favourably in the global analysis in coincidence of species conservation and ecosystem services (a ‘win-win’), within the ecoregion, only ¼ of individual planning units demonstrated a ‘win-win’. Anderson *et al.* (2009) used data at two resolutions 10km x 10km and 2km by 2km. They compared the correlations between biodiversity and 3 other services within the 100km^2 squares with an identical number of randomly sampled 4km^2 cells. The results were very different to the national pattern and could provide completely opposite conclusions in different regions. This led to them suggesting that generalisations were elusive and scale-dependence likely to be a universal influence on the direction and strength of such correlations. Choice of scale should also reflect regional contexts both in ecologocal and policy terms. Priorities will vary regionally to nationally and the beneficiaries of the services will also be different depending on who and at what scale policy is enacted.

Data availability at global and national scales is also an obstacle to quantitative analysis of ecosystem service interactions. Naidoo *et al.* (2008) looked at four services (including biodiversity) for which they were able to get global data and recognised that data availability severely limited their analyses suggesting that more complete analyses using additional services and other aspects of biodiversity might show different results. Chan *et al.* (2006) also stated that the data available for studying ecosystem services is at a very coarse scale and hinders analysis. At a more local scale Raudsepp-Hearne *et al.* (2010) were able to get quantitative data on a larger range of services but stressed that obtaining temporal data collected using consistent methods was virtually impossible.

The extent of the Countryside Survey database and the co-location of many of the variables make it ideal for taking some of these analyses further. The soils and vegetation data are taken within metres of each other thus strengthening the assumption that patterns reflect mechanistic coupling of above and below-ground prcocesses and abiotic conditions. More importantly, it seems likely that service relationships will vary between habitats and CS enables within habitat comparisons of service supply.

Another advantage of the CS dataset is that as well as correlating services spatially it is also possible to investigate correlated patterns of change over time. It should be stressed that it is not the intention to prove causal relationships from these correlations (though some may exist) but rather to determine how services are related to enable better understanding and improved management and to generate further hypotheses that could be more rogorously tested.

Aims and objectives:

1. How are the ecosystem services (including biodiversity) correlated both spatially and temporally?
2. Do these patterns vary with geographic scale and habitat type?
3. How do conservation priorities based on biodiversity capture ecosystem services?

7.2 Methods

Comparing number of services spatially

A simplistic and subjective method of demonstrating spatial variability in ecosystem services can be carried out using expert judgements of the importance of given habitat types for delivering specific ecosystem services. In Britain the results of such a consultation process have been embodied in a matrix of habitat by services (Haines-Young and Potschin 2007). The number of ecosystem services within each habitat was estimated by giving an overall value per habitat (Fig. 7.1a). This can be scaled up to Great Britain using the remotely sensed Land Cover Map 2000. Mapping the total ecosystem service value per Broad Habitat wherever it occurred yields a census map of aggregated service delivery based on expert opinion and where the values within each Broad Habitat remain the same across the country (Fig. 7.1b). This can be combined with maps indicating service demands using human population density and road density (Figs 7.1c and 7.1d). The overlap identifies areas where areas of high ecosystem service delivery may be subject to visitor pressure.

Figure 7.1b concurs with the findings of Anderson *et al.* (2009) that there is a concentration of services in the north and west of Great Britain and less services in the south and east. This is the opposite to the demand maps so there are issues in matching supply and demands, which will need to be considered.

This method has some limitations. The matrix is based on expert judgement so by its nature is subjective, this is particularly apparent in the low number of supporting services included, more likely to be absent because they are less well understood than because they are not present. It is a very crude method of mapping services.

Better understanding of relationships between services can be obtained by using real data collected in Countryside Survey. Maps of individual services can be found within Chapters 3, 4, 5 and 6. As the text explains these are still explorations of the data, however, they enable some visual comparison of spatial relationships.

Simple correlations between Services

Exploration of relationships between services has been carried out using simple correlations between the CS biophysical measurements which have been linked to services in previous chapters of this report. These are shown in Table 7.1.

Table 7.1: Ecosytem service indicator variables analysed.

Ecosystem service	CS metric	Scale
Wild species diversity	Total taxon richness of plant species in CS Main Plots	a, b, c, d,
Wild species diversity	Soil invertebrate diversity	a, b, c, d
Cultural	Appropriate plant diversity	d Can only be done at habitat level
Pollination (regulating)	Bee and Butterfly nectar plant richness	a, b, c, d,
Water Quality (Provisioning/regulating)	Average score per taxon for macro-invertebrates	a, b, c, d, (square level data)
Habitat provision (Provisioning/Biodiversity)	Area of Broad Habitats	a, and selected habitats for b, c,
Topsoil carbon (Regulating-climate)	LOI	a, b, c, d,
Cultural	Charismatic landscapes: Relief, Woods and water, sea and altitude	b, c, d,
Indicator of above-ground Net Primary Production (Supporting)	Cover-weighted Specific Leaf Area	a, b, c, d,

a) GB level data, summary means per habitat, b) high resolution plot/square level data, overall analysis, c) plot/square level data analysed within country, d) plot/square level data analysed within habitat.

The analyses have been carried out at different scales:

a) Great Britain (GB) scale using summary mean values per habitat as produced for the CS UK report[54].
b) These were produced as GB estimates so scaled up statistically from fine scale plot and square level variables.
c) GB; higher resolution, plot and square level values spatial and temporal correlations.
d) Country level using higher resolution plot/square data.
e) Habitat based higher resolution plot/square data.

Table 7.1 indicates which variables have been used for which analysis.

a) GB scale - summary means per habitat

Variables for the GB analysis were slightly different to those from the more detailed analysis. They include total taxon richness of plant species (total taxon number of plant species in a Main Plot), soil invertebrate diversity (total taxa in a core taken from the same plots as the vegetation), bee and butterfly nectar plant richness (see Chapter 5), area of Broad habitat, LOI - topsoil carbon (taken from soil core (0-15cm) in same plot as vegetation and soil invertebrates). Specific Leaf Area (a cover-weighted average of the plants in the Main Plots) is also included as a metric here as an indicator of Specific Above Ground Net Primary Productivity. Also called

[54] http://www.countrysidesurvey.org.uk/reports2007.html

'ecosystem efficiency' SANPP expresses ANPP on a per gram of biomass basis (Garnier *et al.* 2004). Mean values were calculated per habitat as described in the *CS UK results from 2007*[43]. Some measures were not used in this overall analysis; appropriate plant diversity (number of Common Standards Monitoring-CSM- species) as described in Chapter 4 was not suitable as a general measure as it can only be applied at the habitat level. Average Score per Taxon (ASPT) as a measure of stream ecological quality (see Chapter 2) has also not been used nationally as it requires calculation of a mean value per habitat. There is only one freshwater sampling point in a square and although allocation to a habitat could be done using a crude measure such as Broad Habitat with highest proportion in a square it would not be sufficiently accurate. The mean area of a Broad Habitat in GB has been used as a variable.

GB scale correlations were straightforward because they have already been summarised from individual plot and square data to provide means per habitat. They were analysed using the package Statistica 6.0 and the data correlated and r-squared value calculated.

b) GB scale - higher resolution, plot and square level values

These analyses use most of the variables described above, total taxon diversity of plants, total taxa of soil invertebrates, number of bee and Butterfly nectar plants and cover-weighted Specific Leaf Area (SLA). However, Average Score per Taxon (ASPT) was also used as a square level value. The variable OE/ASPT was included and the difference between the two is explained in chapter 2, results can be seen alongside the ASPT results. Rather than using the area of Broad Habitat in GB as a variable, the area of selected habitats within a square was used, these are Broadleaved Woodland, Arable and Improved Grassland. They were selected as habitats that were most likely to have an effect on service interactions but also because they can be related to services themselves, Arable and Improved Grassland are linked to food production and Broadleaved Woodland to timber production and also climate regulation in the form of carbon sequestration and storage. Variables have also been calculated to indicate areas in Entry Level Stewardship agreements (data from Natural England) and areas notified as SSSI's.

The more data rich plot or square level correlations were analysed using the proc mixed procedure in SAS (as referred to in Chapter 4 and in Maskell *et al.* 2009). Rather than type 1 tests, as only two variables are used it is a straightforward regression, doing it using this approach means that auto-correlation of plots grouped within a square can be accounted for. The t value and significance level have been quoted in the results. Analyses have been carried out on the spatial status of habitats in 2007 and on observed change between 1998 and 2007.

c) Country level using higher resolution plot/square data

Variables used as above but analyses carried out within England, Scotland and Wales separately. Only stock analyses were done.

d) Habitat based higher resolution plot/square data

Variables used as for b) but analyses carried out within Broad Habitat. These were classified according to allocation in 2007.

Multi-variate analyses

Multi-variate analyses looking at interactions between several services simultaneously have also been carried out for the GB data and the more detailed plot/square level data.

a) Low resolution average data

Some example spider diagrams have been created to show variations in four services between habitats and in the same habitat over time.

b) High resolution plot/square level

Interactions between multiple services in the more detailed plot/square level dataset have been analysed using ordination (Canoco for windows; ter Braak and Smilauer 2002). Mean values per 1km square were calculated for each service and also standard deviations of the service values. These values were treated as ‘species’ data with the 1km squares as plots. Potential explanatory variables used to constrain the ordination axes were proportion of arable area, membership of Entry Level Stewardship (ELS), proportion of Improved Grassland and SSSI membership. Principle Component Analysis (PCA) of the service indicator by 1km square matrix was carried out first. Explanatory variables were added using redundancy analysis to determine the extent to which they could explain the major gradients of variation in the natrix. Statistical testing was based on forward selection of variables and Monte Carlo permutation tests.

7.3 Results

Results from correlations between services

a) GB scale - summary means per habitat

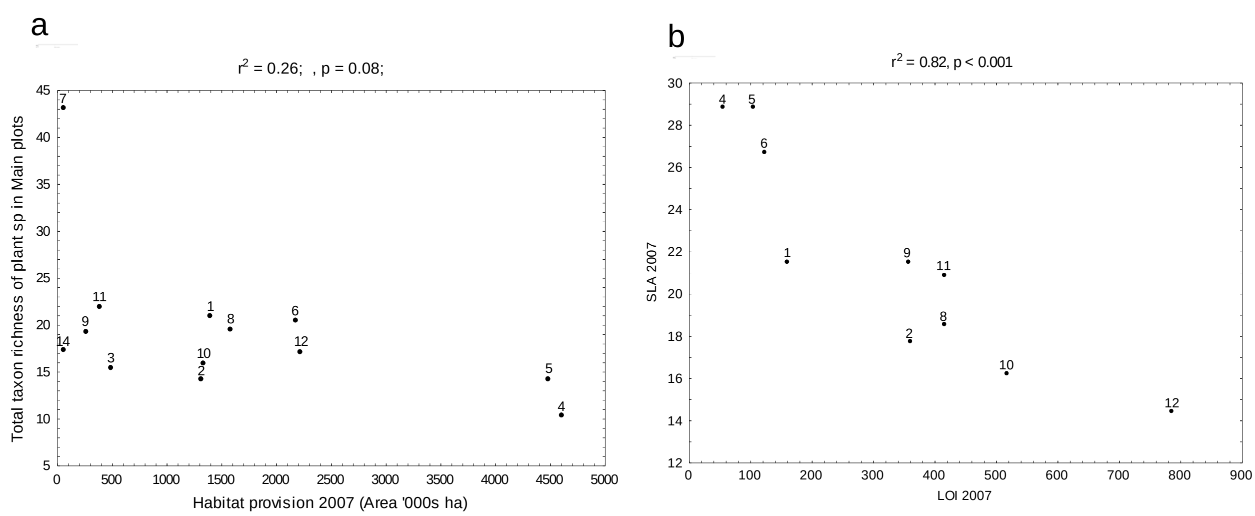
In the more simplistic GB scale analysis (i.e. not plot or square level data) (Appendices; Tables 7.8 and 7.9), there were significant positive relationships between total taxon diversity of plants and butterfly and bee nectar plants. This is to be expected and slightly circular as these plants are a subset of the total taxon richness. Similarly, there is a high correlation between butterfly and bee nectar plants and there will be crossover between them in species composition. Interestingly despite this, in the more detailed results, butterfly and bee nectar plants do not always show the same relationships with other services. There were no significant relationships between soil invertebrate diversity and any other variables which is different to the more detailed analysis.

There was a negative relationship between habitat area in 2007 and plant species richness (Fig. 7.2a). It was not quite significant ($p=0.08$) until Calcareous Grassland Broad Habitat (which has the highest levels of plant richness) was removed and then it became significant (p<0.05). There is a similar relationship between habitat area and butterfly nectar plants (which are a subset of total taxon richness) (r^2=0.35, p<0.05). It is an interesting, although not unexpected, result that the habitats with the largest areas in GB i.e. Arable and Horticulture and Improved Grassland have the lowest diversity (including lowest specialist nectar plants) and that the highest species richness is found in habitats with small total areas, often found as small fragments within the landscape.

There was a significant negative relationship between topsoil carbon (LOI) and Specific Leaf Area (SLA) (Fig. 7.2b.). There were no significant relationships between topsoil carbon (LOI) and species richness. The only significant change result was a positive correlation between change in bee and change in butterfly nectar plants. As mentioned previously there is overlap in species composition between these two groups.

Although there was no overall relationship between a change in habitat area and changes in plant diversity there is a relationship between loss of arable land and increases in plant diversity, CS showed that arable land was converted to other habitats particularly Neutral and Improved Grassland and it is likely this is associated with increases in plant diversity.

Figure 7.2: Results of correlations between mean Broad Habitat values in 2007. a) Habitat provision (habitat area) with total taxon richness of plants, b) LOI (topsoil carbon) with Specific Leaf Area (SLA = surrogate NPP) (1=Broadleaved Woodland, 2=coniferous Woodland, 4=Arable and horticultural, 5=Improved Grassland, 6=Neutral Grassland, 7=Calcareous Grassland, 8=Acid Grassland, 9=Bracken, 10=Dwarf Shrub Heath, 11=Fen, Marsh, Swamp, 12=Bog, 14=streams)



b) GB scale - higher resolution, plot and square level values

Results from the overall plot level analyses of stock in 2007 can be seen in Table 7.2. These results have been summarised in Table 7.4 so detailed reference to them will not be made here. They are discussed further later on. Results for change are presented in Tables 7.3 and summarised in Table 7.6.

c) Plot/square level by country

Results from the individual countries were similar to the GB high resolution results, although there were some differences which are considered further in the discussion section. The fact that there are differences between countries (England, Wales, Scotland and GB) indicates the importance of determining at what scale/management unit trade-offs and bundles of services should be studied as not just the amount of service but the interactions between services will vary. These results can be found in the summary Table 7.4 and more detailed results are in the Appendices; Tables 7.10, 7.11, and 7.12.

d) Within habitat analysis

The results from the stock analyses within habitats are summarised in Table 7.5 and Tables 7.13 to 7.20 in the appendix. The change results are summarised in Table 7.7 and Tables 7.21 to 7.28 in the appendix.

Results from multi-variate analyses

Spider diagrams using GB low resolution data are presented in Figures 7.3a and 7.3b. Figure 7.3a shows a comparison between four services, biodiversity (total taxon richness of plants), habitat provision (habitat area), topsoil carbon (LOI) and butterfly nectar plants in two different habitat types. It demonstrates quite clearly that whilst there are small differences in three of the services, topsoil carbon content is much higher in Bog habitats although plant diversity and nectar plant diversity are slightly lower. Management of an area containing these two habitats to balance service provision would have to consider the increased storage of carbon by Bog and the increased diversity and provision of nectar plants by Neutral Grassland.

Figure 7.3a: Comparison of services between Broad Habitats in 2007.

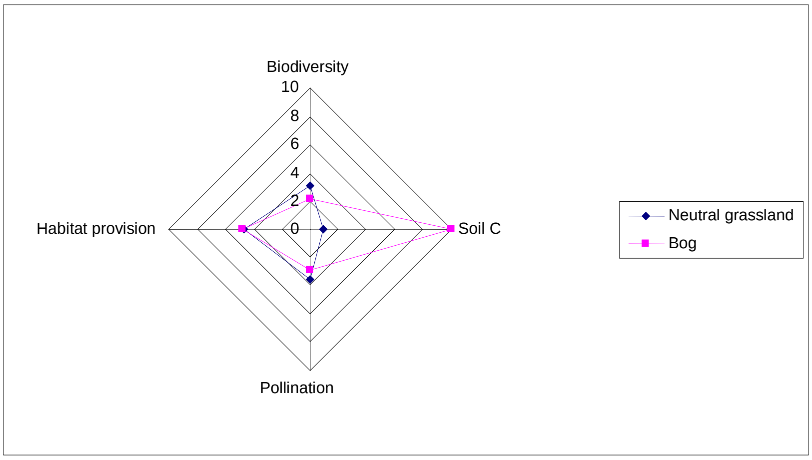
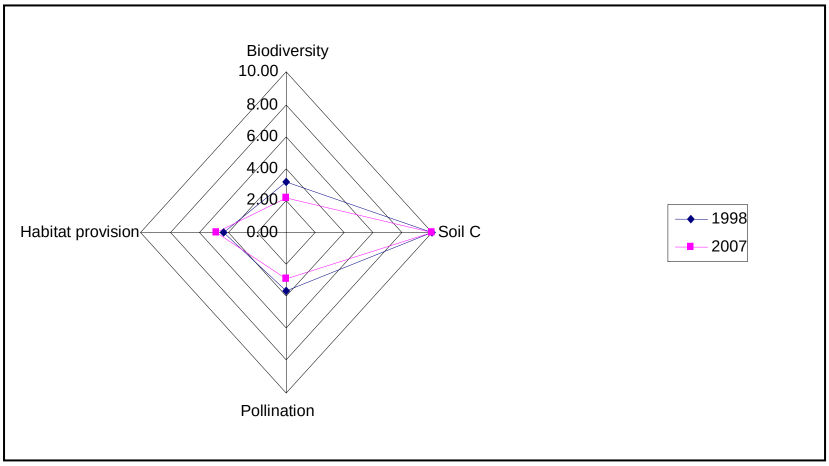


Figure 7.3b shows the differences in services within the Bog Broad Habitat between two time points. This diagram demonstrates that between 1998 and 2007 there were very slight increases in habitat provision and topsoil carbon but the plant diversity and pollination have declined. Overall management of services might consider that the loss of biodiversity and pollination services were compensated for by the increases in topsoil carbon and habitat area.

Figure 7.3b: Comparison of services in the Bog Broad Habitat between 1998 and 2007.



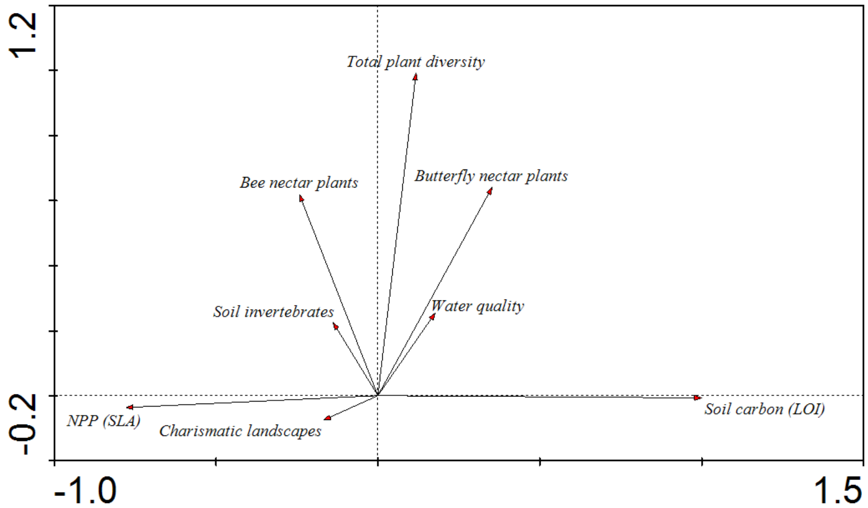
More spider diagrams could be constructed, for other habitats or between different time points, however they are intended here as an example of how to present results for multiple services to visualise the interactions rather than a comprehensive analysis of all possible service and habitat combinations. These diagrams are based on the summary means per habitat which is quite a crude measure. More detailed data-rich multi-variate analyses have been carried out for Figures 7.4a-c and these are a more accurate, quantitative way of representing service interactions.

Figure 7.4a shows the distribution of services across ordination space. It is very interesting that this follows a uni-modal pattern with components of plant species richness in the centre of the hump and topsoil carbon (LOI) and mean cover-weighted SLA at opposing ends of the primary axis.

It appears that the two main gradients here are productivity (X axis) and diversity (Y axis). It is possible that a third gradient when the 3rd axis is plotted is disturbance as soil invertebrate diversity and water quality are on one side and plant and butterfly diversity on the other. Figure 7.4b includes the standard deviations of the variables to demonstrate the variability of services between plots within a square. There is a lot of variation in topsoil carbon. Many soil types are represented across the survey and it seems to be particularly associated with high diversity suggesting that where there is high diversity of habitats and species there is more variation in other services. The cultural service and the stream ecological quality are square level variables so have no within-square variation.

Figure 7.4c shows the services as above and also projects the explanatory variables into ordination space as supplementary variables i.e. they are not constraining the ordination. This is useful because it shows all variables and how they relate to the services and each other even though they may not be significantly associated. This graph shows a clear distinction between the features of a more intensive landscape in the left bottom quadrant, i.e. more Arable and Improved Grassland, more areas in the Entry Level Stewardship (ELS) scheme. At the other end of the gradient are high soil carbon habitats, also some indication that land managed under SSSI is at this end of the gradient, higher stream ecological quality, and Scotland as a country with more semi-natural habitat and a pre-dominance of habitats such as Bog and Heath. Higher species richness particularly plant species richness comes between these two. High carbon habitats and intensively managed nutrient rich habitats both tend to be associated with low diversity either by nutrient limitation (high carbon) or by dominance of competitive plants (high nutrients). This again fits with the unimodal model where intermediate nutrient levels are assocaited with the highest species richness.

Figure 7.4a: Multi-variate analysis (PCA) of services only using Canoco.



CS metric	Service
SLA	NPP
Bee	Pollination
Butterfly	Pollination
Freshwater ASPT	Water quality
LOI	Topsoil carbon
Total plants	Biodiversity (plants)

Figure 7.4b: Multi-variate analysis (PCA) including mean values and standard deviations.

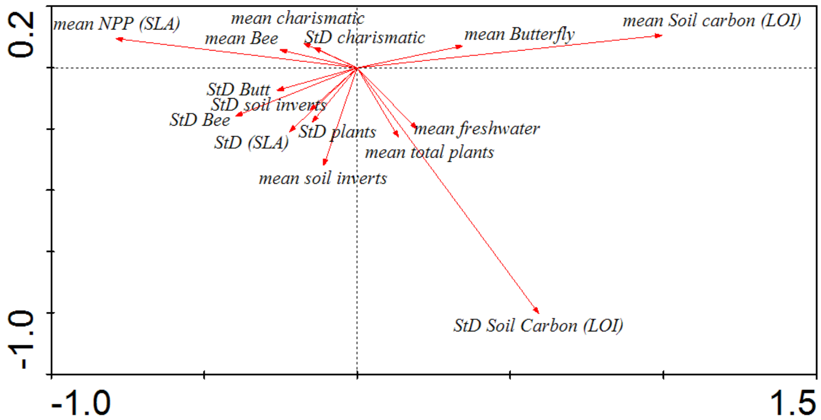
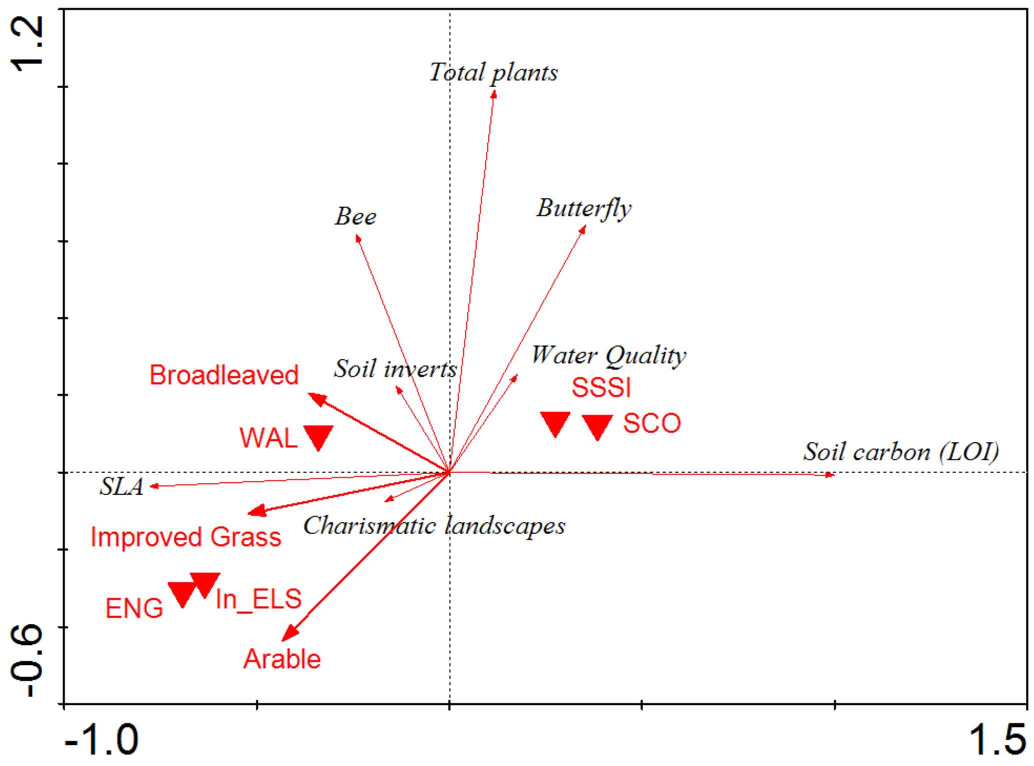


Figure 7.4c: Multi-variate analysis (PCA) of ecosystem service indicators and explanatory variables (red arrows and triangles) added as supplementary variables. (ENG=England, WAL=Wales, SCO=Scotland, SLA= Specific Leaf Area, in_ELS= in Entry Level Stewardship scheme, LOI = loss-on-ignition)



When tested in a constrained ordination (RDA) the variables Improved Grassland, ($F=41.1$, $p<0.001$), Arable area ($F=33.4$, $p<0.001$) and Broadleaved Woodland ($F=17.7$, $p<0.001$) were significantly related to services. Scotland was almost significant ($F=3$, $p=0.06$). This shows statistically that intensive management of habitats has a positive relationship with an indicator of ANPP and a negative relationship with topsoil carbon and water quality (stream ecological quality), pollination (butterfly nectar plants) and plant diversity. Arable land as a variable is directly inversely related to water quality and butterfly pollination.

7.4 Discussion

Spatial relationships

Relationships between diversity components

There are relationships at all scales between the components of plant diversity. In terms of ecosystem services, bee and butterfly nectar plants (pollination), Common Standards Monitoring (CSM) Indicators (appropriate diversity) and total taxon diversity (Biodiversity) play slightly different roles yet there are overlaps in species pool compositions while the first three are subsets of total taxon richness. Bee and butterfly nectar plants are representative of the pollination service yet they do demonstrate different patterns of service relationships and landscape interactions to each other. Total taxon richness represents biodiversity as a service, as a source of genetic material, contributing towards ecosystem resilience and relating to ecosystem function. CSM Indicators represent cultural services. Assemblages of species representative of common habitats are desirable as indicators of favourable habitat condition, although some of these habitats are man-made or maintained by management. In some of the habitats (Acid Grassland and Bog) there was no relationship between CSM Indicators and total taxon diversity of plants, this is where the concept of appropriate diversity is relevant. These are habitats with low nutrient status and low diversity where species richness could be a negative indicator if they are 'undesirable' species i.e. ones associated with eutrophication and disturbance. It is important to take this into account which is why appropriate diversity was developed in Chapter 4 and included here as a response variable in the within-habitat analyses.

There are relationships with taxa other than plants. Although in the low resolution GB analysis there was no relationship with soil diversity, in the higher resolution plot level data there were positive relationships between soil diversity, ASPT stream ecological quality, total plant richness and bee and butterfly nectar plants. It seems very apparent that for these groups where one component of diversity is high it is likely that others will be also. This is also the case within habitats. In Improved Grassland, Neutral Grassland and Acid Grassland there were positive relationships between different biodiversity components including soil diversity and stream ecological quality. There were some negative relationships; in Broadleaved Woodland, soil invertebrates were negatively associated with Ancient Woodland Indicators and total taxon diversity and in Fen, Marsh, Swamp soil invertebrates were negatively associated with bee nectar plants. In Fen, Marsh Swamp it seems possible that richness of bee nectar plants is representing a more managed arable landscape which is having a negative effect on soil diversity. It is difficult to know from this analysis what is happening in the Broadleaved Woodlands, perhaps it is related to disturbance with more soil invertebrates in more disturbed newer woodland possibly as a result of a transition stage where open habitat species are declining and woodland species increasing creating transitory high species diversity.

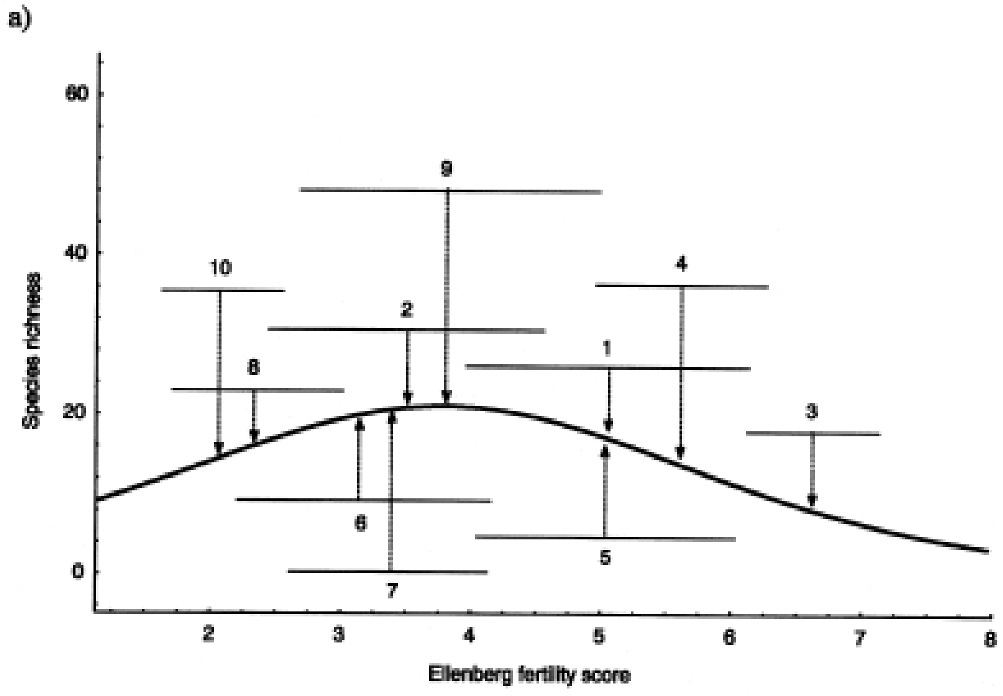
An indicator of Above-ground Net Primary Productivity (Specific Leaf Area) and diversity

Mean cover-weighted Specific Leaf Area (SLA) is being used in this chapter as an indicator of Above-ground Net Primary Productivity (a supporting ecosystem service). SLA has been shown to be related to age and establishment of a community. In the early stages of succession, SLA is high as fast growing species acquiring resources rapidly dominate and as succession proceeds, slower growing species more suited to conservation of resources with a lower SLA increase in number (Garnier *et al.* 2004). It is also strongly correlated with nutrient availability where high SLA species typically have higher relative growth rates and less well defended leaves with higher tissue N content and hence more rapidly decomposable litter (Díaz et al 2004; Wilson et al 1999).

One of the most obvious differences in relationships to productivity (SLA) is between bee and butterfly nectar plants. Both are being used as indicators of pollination but bees are positively related to mean cover-weighted SLA and butterflies negatively related. This seems to be because the most common nectar providing plants for bees are nutrient-demanding nitrophiles while nectar plants for butterflies are more commonly assocaited with less productive situations (see Appendix 5.1).

Relationships between mean cover-weighted SLA and diversity vary amongst habitat types, positive relationships between plant diversity and mean cover-weighted SLA exist in Broadleaved Woodland (although not CSM Indicators), Acid Grassland, Bog and Dwarf Shrub Heath. Acid Grassland and Dwarf Shrub Heath are low nutrient habitats to the left of a unimodal curve so an increase in resource availability has the potential to increase species diversity. Figure 7.5 taken from (Smart *et al.* 2003) shows a similar relationship between mean Ellenberg fertility and species richness with the Broad Habitats in Countryside Survey in 1998 positioned along the curve. This demonstrates clearly how the relationship between species diversity and productivity varies with habitat. Species diversity of habitats already rich in nutrients such as Arable, Improved Grassland and Neutral Grassland are negatively related to mean cover-weighted SLA so as productivity increases diversity goes down. Broadleaved Woodland is slightly anomalous. Fen, Marsh, Swamp shown in the centre of the graph on the peak is similarly poised with mean cover-weighted SLA. Bee nectar plants are positively related to SLA and soil invertebrates negatively associated with mean cover-weighted SLA.

Figure 7.5: The location of British Broad habitats along an indirectly measured environmental gradient of Ellenberg Fertility versus species richness. The position of the mean Ellenberg value among plants in each Broad Habitat is shown plus and minus the standard deviation about the mean. Codes for Broad Habitats; 1. Broadleaved Woodland, 2. Coniferous Woodland, 3. Arable and Horticulture 4. Improved Grassland, 5. Neutral Grassland, 6. Acid Grassland, 7. Bracken, 8. Dwarf Shrub Heath, 9. Fen, Marsh, Swamp, 10. Bog (taken from Smart et al. 2003).



Topsoil carbon (LOI in 0-15cm) and diversity

Although total plant diversity is not correlated with topsoil carbon at the GB scale there do seem to be some relationships between topsoil carbon and diversity. Bee nectar plants are negatively related to soil carbon at the scale of GB and also within habitats (Neutral Grassland, Acid Grassland and Fen, Marsh, Swamp), perhaps because they tend to be found in more agricultural landscapes which tend to contain less carbon so where carbon is higher the habitats are likely to be less managed. Butterfly nectar plants on the other hand tend to be positively associated with topsoil carbon in GB (except in Acid Grassland) (Table 7.2). This may be because higher topsoil carbon is associated with less intensive habitats favouring butterfly nectar plants. There are differences in the relationship between countries, in England higher topsoil carbon is associated with higher plant and soil invertebrate diversity whereas in Scotland the opposite pattern can be seen. This is perhaps because in England higher carbon indicates less agricultural soils and is associated with increases in diversity, whereas in Scotland carbon-rich habitats such as bog, plant diversity tends to be lower, soils are waterlogged and decomposition processes slower. This apparent disparity in country results when it is actually different habitats and associated habitat properties that are being measured, makes the point that it is important to compare like with like and within habitat analyses are important.

Within some habitats (Acid Grassland, Fen, Marsh, Swamp and Bog) topsoil carbon (LOI) is negatively related to plant diversity. This is not appropriate diversity measured by CSM Indicators but is total plant diversity which could be an indication of poor condition so increased carbon could still be related to favourable condition. Only in Improved Grassland was there a positive relationship between topsoil carbon and plant diversity but here increased diversity is a desired outcome.

Water quality and other services

ASPT (Average Score per Taxon for freshwater macro-invertebrates) was used as a measure of freshwater ecological condition, it is used as a standard indicator of water quality by UK monitoring agencies. Chapter 2 discusses the merits of using RIVPACS adjusted O/E ASPT which removes any underlying spatial pattern in biotic indices, it was thought that it might be more appropriate given the nature of the other variables to use the raw ASPT scores here but as shown in Tables 7.2 and 7.3, the results from OE/ASPT were included and they show similar patterns. As mentioned above ASPT (freshwater ecological condition) is positively related to plant and soil diversity at the GB scale, this includes butterfly nectar plants but there is no significant relationship with bee nectar plants.

Freshwater ecological condition is negatively related to mean cover-weighted Specific Leaf Area (SLA) reflecting a negative association with the most productive ecosystems. Freshwater ecological condition is positively related to topsoil carbon (at the GB scale and in England but not significant within habitats). This may be because higher topsoil carbon is associated with less intensive habitats where water quality is higher. Few of the within habitat analyses were significant but there was a positive relationship between ASPT and bee and butterfly nectar plants in Improved Grassland. Higher diversity in this intensive habitat possibly associated with less intensive management also seems to be reflected in better stream condition. There was also a similar relationship in Broadleaved Woodland but this time with changes in bee and butterfly nectar plants and freshwater quality. In Dwarf Shrub Heath a negative relationship between freshwater ecological quality and total plant species richness may signify a loss of condition resulting in not only unfavourable vegetation status but also poor freshwater quality.

Net Primary Productivity (Specific Leaf Area) and topsoil carbon (LOI)

A negative relationship between mean cover-weighted SLA and topsoil carbon was apparent in most analyses, whatever the scale and within most habitats (only within the arable habitat was this relationship not apparent). This is very interesting and demonstrates an inverse relationship between above-ground and below-ground processes. A negative relationship between mean cover-weighted SLA and topsoil carbon is not unexpected, Garnier *et al.* (2004) found a significant relationship (-0.88) between mean SLA and topsoil carbon. Fast growing species with high SLA producing high quality litter tend to have soils with lower carbon. Fast growing species are replaced by slower growing species with lower SLA that decompose more slowly as a community becomes more established, so topsoil carbon builds up in communities with low SLA. Conditions inhibiting decomposition and hence promoting build-up of soil carbon are known to be associated with habitats with low ANPP. This significant relationship gives a means of estimating topsoil carbon from a vegetation property, however, it doesn't account for situations where above-ground and below-ground processes are in disequilibrium e.g. the fens where growing arable crops will result in high SLA whilst the topsoil carbon is still high. Where it could be useful is in optimising carbon sequestration; habitats currently high in soil carbon tend to be inhabited by slow-growing species with low SLA. If a balance could be achieved between topsoil carbon levels and more productive species, carbon sequestration could be optimised. This might be most achievable in habitats such as Neutral and Acid Grassland where topsoil carbon levels are intermediate.

Intensification and other services

This analysis has not measured food production quantitatively. Obtaining data on yield, crop production and stocking was attempted but it was not available for the whole of GB at a fine enough resolution to be able to match yield data with ppolygons conatining sampling plots possible at the GB scale. Instead the areas of Arable and Improved Grassland within a square were used to signify food production and evaluate relationships with other services. This is a key area where the concept of 'trade-offs' comes into play. Arable land is associated with low species diversity of all types, total plants, pollinator plants, soil and stream ecological quality and low carbon. Improved Grassland demonstrates a positive relationship with bee nectar plants but negative relationships with freshwater ecological quality and topsoil carbon. Mean cover-weighted SLA productivity is positively related to arable area and if we had a metric for food production this would be high in these areas and it might well be necessary to prioritise food production at the expense of other services in some areas. This was the case at the GB scale and in all three devolved countries so it is a policy issue for all. This conflict between production and other services was found in the analysis by Raudsepp-Hearne *et al.* (2010). The two most intensively managed areas (pork production and crop production) had the highest number of negative relationships with other services. The loss of regulating and cultural services associated with these areas of high provision may have implications for the sustainability of these areas for food production as well as being costly for the farmer to replenish soil properties using artificial fertilisers, so it is not just a case of choosing food over other services.

The CS analysis is based on Main Plots so it doesn't allow for the retention of diversity in refuges such as small habitat patches, field margins and hedgerows which are more likely to be targeted for conservation measures. Unfortunately, even in some of these (e.g. small habitat patches) diversity has been declining (UK Results from 2007, Carey et al. 2008). The CS results for the UK also showed an increase in plant diversity in arable Main Plots between 1998 and 2007 so diversity can be improved. However, arable habitats tended to have lower intrinsic species richness than other semi-natural habitats. A preliminary analysis of ELS in England and within arable, Improved and Neutral Grassland habitats was included in this analysis. It showed negative results between Entry Level Stewardship (ELS) membership and plant diversity (total plants, bee and butterfly nectar plants) and stream ecological quality. Since the ELS scheme was only launched in 2005, it may have been too soon for measurements made in CS in 2007 to detect the consequences of changed management. In addition, there are different options that can be applied and this analysis used a crude measure of whether a plot was in scheme or out of scheme based on holding level data.

Figures 7.2a and 7.2b show low diversity habitats cover the largest proportion of the country. If even small changes could be made to increase the topsoil carbon content, pollination and water purification properties of Arable land and Improved Grassland, then the net result could be quite large. However, this should be an additional measure for gains in service provision, not instead of conservation on special sites. The species gained in the wider countryside are not comparable to those from special sites and it is important to ensure that sites where services are currently high are not degraded.

Proportion of woodland area in a 1km square

Woodland area in a square was positively related to species richness in GB, England and Wales and to mean cover-weighted SLA in GB and Scotland. It was negatively related to topsoil carbon, but in woodlands carbon is sequestered into the above-ground biomass.

Protected areas

Designation as SSSI was associated with high carbon habitats. There was also a positive association with soil invertebrate diversity. There were no significant positive associations with components of plant diversity and even some negative ones (including with CSM Indicators) in Dwarf Shrub Heath. Surprisingly, there were no positive relationships between SSSI designation and numbers of positive CSM Indicators. SSSI designation and subsequent management is very specific for the type of habitat. A more complex analysis that determined more accurately what prescriptions were applied to a particular piece of land might tease out more results. Bee nectar plants were negatively associated with SSSI designation, possibly for reasons suggested above that they tend to be more associated with arable, lowland habitats.

Naidoo *et al.* (2008) showed variations in services between different types of conservation areas, high-biodiversity wilderness areas were better for carbon storage and sequestration than biodiversity hotspots but worse for grassland production. Perhaps the high-diversity wilderness areas of Naidoo’s study are more equivalent to SSSI’s where the larger areas are carbon rich habitats (such as bogs and moorland) which are less productive for food yet moderately rich in diversity.

‘Charismatic landscapes’ and other services

The cultural service used for these analyses was *‘Charismatic landscapes’* Cultural Service 1 (see Chapter 6) - a value comprised of data on woodlands, water, sea, altitude and relief for a CS square (these being components of a culturally valued landscape). This seems to be consistent with high values of other services, plant species richness, butterfly nectar plants (in GB and England not Scotland), freshwater ecological quality and soil invertebrate diversity. There was a negative relationship with bee nectar plants (in GB and Scotland), perhaps because the habitat components favoured tend to be more representative of upland or woodland areas or at least squares containing more semi-natural habitat and less likely to contain bee nectar plants. The cultural service was also associated with higher topsoil carbon (LOI) and lower productivity (SLA), possibly because the highest densities of water bodies and streams and greatest variation in relief (a major component of this cultural service) are found in the uplands where soil carbon is highest and annual plant production lowest.

Protection (SSSI)

In areas notified as SSSI there were less bee nectar plants, more topsoil carbon (LOI) and lower SLA, again these are the opposite patterns to those of more intensive land uses and probably the result of the different types of habitats likely to be notified as SSSI (e.g. higher carbon habitats such as bogs and heathlands). In England there was a positive association between SSSI designation and soil invertebrate diversity not detected elsewhere. There was a positive relationship between SSSI designation and soil invertebrate diversity in Dwarf Shrub Heath, Fen, Marsh, Swamp and Neutral Grassland. There were negative results between SSSI designation and total plant diversity in Dwarf Shrub Heath and Bog and with CSM Indicators and bee and butterfly nectar plants in Dwarf Shrub Heath. In Fen, Marsh, Swamp the negative results were with bees only.

Scale

Although there were similarities between GB scale results and those from the individual countries (e.g. negative relationship between water quality, topsoil carbon and SLA in all 3 countries and GB) there were also differences which reflect the types of habitat, landuse, land management and policy applied in these countries. Topsoil carbon (LOI) was positively related to plant and soil invertebrate species richness in England but negative in Scotland and Wales. This was also the case for relationships between plant and soil diversity. This is probably a reflection of more arable, resource rich habitats in England and high carbon habitats that are nutrient limited but could still be in favourable condition for appropriate diversity in Scotland and Wales.

Soil invertebrates were negatively related to SLA in England and positively related to SLA in Scotland. Plant and soil diversity, bee and butterfly nectar plants were all positively correlated with water quality in England. Only bee nectar plants were related to water quality in Scotland and this was a negative relationship.

Analyses within habitats show even more variation reflecting the different ecosystem processes that operate at this scale.

Temporal relationships

In general there were less significant temporal relationships than spatial.

Changes in diversity components

In most habitats changes in one component of plant species richness was positively related to another. As mentioned above it is likely that factors that influence richness in one aspect of diversity have favourable impacts on the others. In arable habitats as well as changes in plant species richness components (including the arable dicots analysed as CSM Indicators) being positively related there was also nearly a significant relationship with change in soil invertebrate diversity. Different factors are operating in different habitats. In Arable and Horticulture it could be less intensive management allowing more species to co-exist, in Acid Grassland increased grazing could be the factor favouring species richness. In some habitats such as Bog, the positive relationships in diversity between pollination indicators and total plants does not extend to CSM Indicators. It could be that undesirable species for that habitat type are encroaching which might also be indicated by the negative relationship between soil invertebrates and butterfly nectar plants. Stream ecological quality and total plant diversity in Dwarf Shrub Heath were negatively related, again in this case increases in plant diversity might indicate negative condition as they represent a move away from appropriate diversity levels.

Changes in Net Primary Productivity (Specific Leaf Area) and diversity

In the GB (high resolution) analysis changes over time in butterfly nectar plants are significantly negatively correlated with changes in Specific Leaf Area i.e. as SLA (productivity) increases butterfly nectar plants decline. This corresponds to the results from the stock analyses where butterfly nectar plants are associated with high carbon and low productivity. Within habitats changes in NPP (SLA) have different effects and implications according to habitat type. In Arable and Horticulture Broad Habitats increases in SLA, signifying increases in productivity, reduce plant diversity. Soil invertebrate diversity in this Broad Habitat is also almost significantly negatively related to SLA. In Acid Grassland and Bogs, increases in NPP are positively related with some aspects of diversity (total plants and bee and butterfly nectar plants) but not CSM Indicators. Increased diversity may be due to increases in nutrient availability enabling colonisation by mesotrophic species, but where diversity is increased the species may not be desirable or representative of the habitats. In Broadleaved Woodland increased productivity is positive for total plant diversity and Ancient Woodland Indicators but negative for soil invertebrate diversity.

Changes in topsoil carbon (LOI) and diversity

Increases in topsoil carbon were positively related to changes in plant diversity in Arable habitats which may be linked to an increase in the amount or quality of carbon inputs and/or a slowing of decomposition rates associated with the change in quality of plant litter with a more diverse flora. However, in Neutral Grasslands a negative relationship was observed which cautions against generalisations about the link between the potential benefits of increased plant diversity for soil carbon sequestration. No significant relationships were identified for other habitats. It is perhaps noteworthy that it is two habitats with lower topsoil carbon concentration where patterns have been observed. Significant changes can be more easily identified when the pool size is small.

In Acid Grassland and Dwarf Shrub Heath increases in soil carbon were correlated with increases in soil invertebrate diversity. Soil invertebrate diversity is often linked to resilience of soil function rather than absolute abundance. In addition, an increase in topsoil carbon may also be linked to a slowing of decomposition rates providing an example of how a slowing of an ecosystem process leads to an increase in an ecosystem service. In the Bog Broad Habitat, increases in topsoil carbon and butterfly nectar plants are positively related. No obvious rationale for this is apparent.

Changes in water quality

There were no significant changes over time between water quality and other services at the GB scale. Within habitats there was a positive relationship in Broadleaved woodland between changes in bee and butterfly nectar plants and changes in freshwater quality. There was a negative relationship between changes in stream ecological quality and changes in total plant diversity in Dwarf Shrub Heath. In this low nutrient, low diversity habitat higher species richness can be a negative indicator of condition. There were no significant relationships between change in stream ecological quality and change in mean cover-weighted SLA or change in topsoil carbon.

Changes in mean cover-weighted Specific Leaf Area and topsoil carbon (LOI)

Despite the strong spatial relationship shown between Specific Leaf Area and topsoil carbon there were no significant temporal relationships.

Change in arable area and other services

Change in arable area is significantly negatively correlated with plant species richness i.e. when arable land increases diversity decreases, this includes all species so not just the effect on arable dicots. This is consistent with the trends in stock discussed above where arable area is associated with lower plant diversity and water quality. There is a positive association of changes in arable area with soil diversity which is perhaps more difficult to explain. Perhaps there is a time lag in changes to soil invertebrate diversity.

Changes in woodland area

There also seem to be a loss of plant species with increases in woodland area (apart from butterfly nectar plants which increased). Possibly due to the loss of species from open habitats as canopy cover increases, which tends to limit the number of dominant species.

Multi-variate analyses of ecosystem services

The multi-variate analyses are much more useful than correlations between two variables for understanding complex relationships between services within a landscape. These understandings are likely to be critically important for the management of bundles of ecosystem services at specific localities. Methods are being developed for multi-variate service analysis. Tools such as InVest (Integrated Valuation of Ecosystem Services and Tradeoffs), which is a spatially explicit modelling tool based on ecological production functions and economic valuation (Nelson *et al.* 2009) has been used as an exemplar to look at services on a relatively local scale (in Hawaii). This study used input data very similar to CS representing services e.g. carbon storage, water quality, biodiversity and a base land Cover Map and combined this with scenarios. Ordination is another tool that can be used effectively as we have done here to look at not only relationships between services but how they relate to external drivers. There are several important conclusions from our analyses:

a) The two main gradients identified operating across all the services studied were productivity and diversity. Productivity and high topsoil carbon were at opposite ends of the gradient. It is the habitats in the middle of the gradient with intermediate diversity that are more susceptible to changes in management to benefit more than one service. Also, managing an area for services requires managing the composition of habitats within that area rather than each habitat individually.
b) Intensive management of habitats an expectedly positive relationship with productivity (indicated by mean cover-weighted SLA) and a negative relationship with topsoil carbon and water quality (stream ecological quality), pollination (butterfly nectar plants) and plant diversity.
c) The 1km squares with the highest species diversity also had the highest variation in other services such as topsoil carbon. This suggests that high habitat heterogeneity i.e. diversity of both habitats and species will maximise services.
d) The ordination models identifying service interactions can be used to predict the values of ecosystem service indicators in relation to the proportion of Broad Habitats within a 1km square. This could then be extended beyond CS squares to all 1km squares in Britain using the composition of Broad Habitats taken from the new Land Cover Map for 2007.

7.5 Conclusions and further work

Study of the interactions between ecosystem services is a developing area of science and this chapter provides an exploration of core CS data that can help to understand and quantify the spatial and temporal correlations between ecosystem services and biodiversity. The work has produced some extremely interesting results, raises many questions and suggests a number of hypotheses. Some of the relationships demonstrated are unsurprising, for example the negative relationship between mean cover-weighted Specific Leaf Area (SLA) and topsoil carbon (LOI) yet nonetheless have not often been demonstrated at the large-scale across a random, representative sample of ecosystems. The variation in results between GB, country and habitat makes it apparent that the direction and strength of ecosystem service correlations is critically scale-dependent.

The use of multi-variate techniques in the study of service interactions is fairly novel particularly at this scale and there is much further work that could be done. Some would be relatively straightforward e.g. including additional explanatory variables and some more complex such as using the relationships calculated from the ordination to extrapolate from CS squares to the wider countryside in association with mapping techniques.

This study was restricted to using quantitative data from CS to represent services; it would be interesting to compare CS data and results with other datasets providing biophysical measures. For instance it is very likely that different measures of diversity will have different relationships with services. This is the case to some extent with soil invertebrates as compared to plant diversity. A better understanding of how a range of diversity measures interact with other services is desirable but will require linkage with other datasets.

Chapter 7: On-line appendices

7.1 Correlation matrices and significance test results for ecosystem service indicators.

Table 7.2: Results from correlations of plot/square level variables, stock in 2007, *p<0.05, **p<0.01, ***p<0.001

	Plant species richness	Soil invertebrate diversity	Bee nectar plants	Butterfly nectar plants	ASPT stream ecological quality	OE/ASPT	LOI	Specific Leaf Area (SLA)
Plant species richness		+3.1**	+35.6**	+42.2***	+3.9***	2.8**	ns	ns
Soil invertebrate diversity			+3.1**	+3.2***	+2.6**	ns	ns	ns
Bee nectar plants				+49.2***	ns	ns	-7.2***	+9.7***
Butterfly nectar plants					+4.4***	4***	+4.2***	-4.7***
Average Score Per Taxon (ASPT) stream ecological quality							4.4***	-5.9***
Observed to Expected ratio (OE) /ASPT							3.5***	-4.2***
LOI (topsoil carbon)								-21.9***
cultural	+3.9***	+4.9***	-2.4*	+2.1*	ns	3.9***	+7.1***	-10.2***
Arable area	-11.4***	-3.6***	-7.6***	-11.4***	-6.8***	-4.3***	-11.4***	+12***
Woodland area	+5.3***	+5.2***	+5.9***	+3.7***	ns	ns	-6.6***	+3.2***
Improved Grass area	ns	ns	+8.4***	-2*	-5.1***	-3.8***	-11.5***	+15.6***
Site of Special Scientific Interest (SSSI)	ns	ns	-3.7***	ns	2.5*	ns	+5.4***	-10.1***

Table 7.3: Results from correlations of plot/square level variables, change between 1998 and 2007, *p<0.05, **p<0.01, ***p<0.001

	Change in plant species richness	Change in Soil invertebrate diversity	Change in Bee Nectar plants	Change in Butterfly Nectar plants	Change in ASPT stream quality (Observed to Expected ratio (OE))/ASPT also ns)	Change in LOI	Change in Specific Leaf Area (SLA)
Change in plant species richness		ns	+25***	+24***	ns	ns	ns
Change in soil invertebrate diversity			ns	ns	ns	ns	ns
Change in Bee Nectar plants				+48.8***	ns	ns	ns
Change in Butterfly Nectar plants					ns	ns	-3*
Change in Average Score Per Taxon (ASPT) stream ecological quality						ns	ns
Change in LOI (topsoil carbon)							ns
Change in Arable area	-7.2***	+2.4*	ns	ns	ns	-2*	ns
Change in Woodland	-2.6*	ns	ns	+6.5*	ns	ns	ns
Change in Improved Grassland	ns	ns	ns	ns	ns	-1.7 p=0.08	ns

Table 7.4: Summary of stock results from Chapter 7, green indicates positive relationships between services (synergies), red- negative relationships between services (trade-offs) and amber a mixture of positive and negative relationships.

Service	CS metric	GB	Plot level GB	England	Scotland	Wales
Wild species diversity	Plant and soil diversity relationships	+ve bee, butterfly and total plants	+ve all	+ve all	+ve bee, butterfly and plant species richness only	+ve bee, butterfly and plant species richness only
Wild species diversity and NPP	Plant and soil diversity and SLA	ns	+ve bee -ve butterfly	+ve bee -ve soil inverts, butterfly	+ve bee, soil inverts -ve butterfly	+ve bee
Wild species diversity and topsoil carbon	Plant and soil diversity and topsoil carbon (LOI)	ns	+ve butterfly -ve bee nectar plants	+ve plant, soil inverts, -ve bee	+ve butterfly -ve plants, soil inverts, bee	-ve plants, bee, butterfly
Water purification and wild species diversity	Water quality (ASPT) and diversity of plants and soil	na	+ve plant and soil diversity, butterfly bee ns	+ve all	-ve bee	ns
Water purification and SLA	Water quality (ASPT) and SLA	na	-ve	-ve	-ve	-ve
Water purification and topsoil carbon	Water quality (ASPT) and topsoil carbon (LOI)	na	+ve	+ve	ns	ns
Topsoil carbon and NPP	Topsoil carbon (LOI) and SLA	-ve	-ve	-ve	-ve	-ve
Food productivity and other services	Proportion of Arable area in a 1km square	na	+ve SLA -ve plant and soil species richness, water quality, LOI, bee	+ve SLA -ve plant and soil species richness, water quality, LOI, bee	+ve SLA -ve plants, bee, butterfly, water quality, LOI	+ve SLA -ve plants, LOI
Cultural services	Charismatic landscapes and other services	na	+ve plant spp, soil inverts, butterfly, water quality, LOI -ve bee, SLA	+ve plants, soil inverts, butterfly, LOI -ve SLA	+ve soil inverts -ve bee, butterfly, SLA	+ve LOI -SLA
Cultural services	Protection (Site of Special Scientific Interest, SSSI)	na	+ve water quality -ve bee	+ve soil, LOI -ve bee, SLA	+ve LOI -SLA	+ve LOI -ve bee, butterfly, SLA

[ASPT = Average Score per Taxon; Bee = Bee nectar plants; Butterfly = Butterfly larval food plants; LOI = Loss on Ignition (topsoil carbon); NPP = Net Primary Productivity; SLA = mean cover-weighted Specific Leaf Area]

Table 7.5: Summary of stock results from Chapter 7 within habitat, green indicates positive relationships between services (synergies), red-negative relationships between services (trade-offs) and amber a mixture of positive and negative relationships.

Service		Broadleaved Woodland	Arable	Improved Grassland	Neutral Grassland	Acid Grassland	Dwarf Shrub Heath	Fen, Marsh and Swamp	Bog
Wild species diversity	Plant and soil diversity relationships	+total plants, AWI, bee, butterfly, -ve soil inverts, total plants, AWI	+ CSM, total plants, bee, butterfly	+ve ASPT, bee, butterfly, total plants	+ve total plants, CSM, bee, butterfly, soil inverts	+ve total plants, CSM, bee, butterfly, soil inverts	+ve total plants, CSM, bee, butterfly	+ve total plants, CSM, bee, butterfly -ve bee, soil inverts	+ve total plants, bee and butterfly NOT CSM
Wild species diversity and NPP	Plant and soil diversity and SLA	+ve total plants, bee, butterfly	-ve total plants, CSM, bee, butterfly, soil inverts	-ve total plants, bee, butterfly	-ve total plants, butterfly	+ve total plants, bee, butterfly	+ve total plants, butterfly	+ve bee -ve soil inverts	+ve total plants, soil inverts, butterfly
Wild species diversity and topsoil carbon	Plant and soil diversity and Topsoil carbon (LOI)	ns	ns	+ve total plants	-ve bee	-ve total plants, bee, butterfly	ns	-ve total plants, bee	-ve total plants
Water purification and wild species diversity	Water quality (ASPT), diversity of plants and soil	ns	ns	+ve bee, butterfly	ns	ns	ns	ns	ns
Water purification and SLA	Water quality (ASPT) and SLA	ns	ns	ns	ns	ns	ns	ns	ns
Water purification and topsoil carbon	Water quality (ASPT) and LOI	ns	ns	ns	ns	ns	ns	ns	ns
Topsoil carbon and NPP	Topsoil carbon (LOI) and SLA	-ve	ns	-ve	-ve	-ve	-ve	-ve	ns
Cultural services	Protection (Site of Special Scientific Interest, SSSI)	ns	ns	ns	+ve soil inverts	ns	+ve soil inverts -ve total plants, CSM, bee, butterfly	+ve soil inverts, -ve bee, SLA	-ve total plants

[ASPT = Average Score per Taxon; AWI =Ancient Woodland Indicator; Bee = Bee nectar plants; Butterfly = Butterfly larval food plants; CSM = Common Standards Monitoring; LOI = Loss on Ignition (topsoil carbon); NPP = Net Primary Productivity; SLA = mean cover-weighted Specific Leaf Area]

Table 7.6: Summary of change results from Chapter 7, green indicates positive relationships between services (synergies), red- negative relationships between services (trade-offs) and amber a mixture of positive and negative relationships.

Change in Services	Change in CS Metric	GB	Plot level overall
Change in wild species diversity	Change in plant and soil diversity relationships	+ butterfly and bee	+ve total plants, bee, butterfly
Change in wild species diversity and NPP	Change in plant and soil diversity and SLA	ns	-ve butterfly
Change in wild species diversity and topsoil carbon	Change in plant and topsoil diversity and topsoil carbon (LOI)	ns	Ns
Change in water purification and wild species diversity	Change in water quality (ASPT) and diversity of plants and soil	na	Ns
Change in water purification and SLA	Change in water quality (ASPT) and SLA	na	Ns
Change in water purification and topsoil carbon	Change in water quality (ASPT) and LOI	na	Ns
Change in topsoil carbon and NPP	Change in topsoil carbon (LOI) and change in NPP (SLA)	ns	Ns
Change in food, productivity	Change in Arable area	na	+ve soil inverts -ve total plants, LOI

[ASPT = Average Score per Taxon; Bee = Bee nectar plants; Butterfly = Butterfly larval food plants; LOI = Loss on Ignition (topsoil carbon); NPP = Net Primary Productivity; SLA = mean cover-weighted Specific Leaf Area]

Table 7.7: Summary of change results from Chapter 7 within habitat, green indicates positive relationships between services (synergies), red-negative relationships between services (trade-offs) and amber a mixture of positive and negative relationships.

Change in Services	Change in CS metrics	Broadleaved Woodland	Arable	Improved Grassland	Neutral Grassland	Acid Grassland	Dwarf Shrub Heath	Fen Marsh Swamp	Bog
Change in wild species diversity	Change in plant and soil diversity relationships	+total plants, AWI, bee, butterfly	+ CSM, plants, bee, butterfly	+ve bee and butterfly	+ve total plants, CSM, bee, butterfly	+ve total plants, CSM, bee, butterfly	+ve total plants, CSM, bee, butterfly	+ve total plants, CSM, bee, butterfly	+ve total plants, bee and butterfly NOT CSM -ve soil inverts with butterfly
Change in wild species diversity and NPP	Change in plant and soil diversity and productivity SLA	+ve total plants, CSM -ve soil inverts	-ve total plants, CSM, bee, butterfly, soil inverts nearly	ns	ns	+ve bee, butterfly	ns	ns	+ve total plants
Change in wild species diversity and topsoil carbon	Change in plant and soil diversity and topsoil carbon (LOI)	ns	+ve total plants	ns	-ve total plants	+ve soil inverts	+ve soil inverts	ns	+ve butterfly
Change in water purification and wild species diversity	Change in water quality (ASPT) and diversity of plants and soil	+ve bee, butterfly	ns	ns	ns	ns	-ve water quality with total plants	ns	ns
Change in water purification and SLA	Change in Water quality (ASPT) and SLA	ns	ns	ns	ns	ns	ns	ns	ns
Change in water purification and topsoil carbon	Change in water quality (ASPT) and LOI	ns	ns	ns	ns	ns	ns	ns	ns
Change in topsoil carbon and NPP	Change in topsoil carbon (LOI) and change in NPP (SLA)	ns	ns	ns	Ns	ns	ns	ns	ns

[ASPT = Average Score per Taxon; AWI = Ancient Woodland Indicator; Bee = Bee nectar plants; Butterfly = Butterfly larvae food plants; CSM = Common Standards Monitoring; LOI = Loss on Ignition (topsoil carbon); NPP = Net Primary Productivity; SLA = Specific Leaf Area]

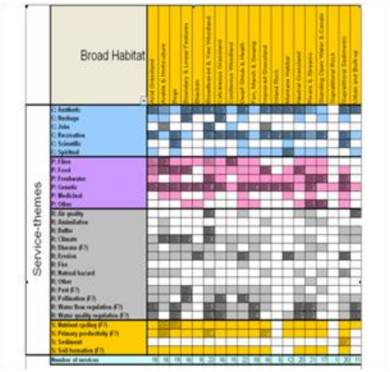


Figure 7.1a: Matrix shows the association between ecosystem services and the BH's in England, (table based on expert consultations). The cross-hatched cells are those identified as most important services associated with each BH. Taken from Haines-Young (2007).

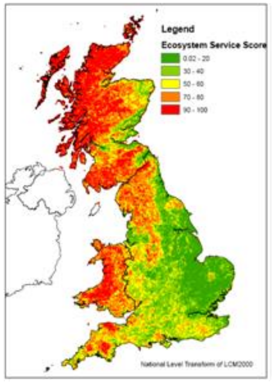


Figure 7.1b: Realisation of matrix, values per habitat scaled up be Landclass means.

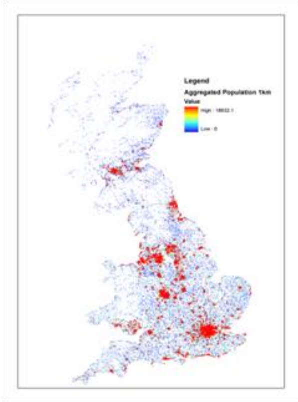


Figure 7.1c: Population density map of Great Britain.

Figure 7.1d: Road density network.

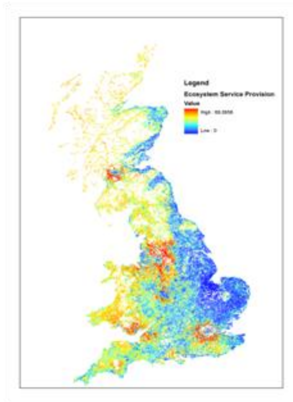


Figure 7.1e: Map combines maps b, c and d to identify regions of high ES delivery subject to potentially high visitor pressure.

Chapter 8: Some policy implications of the Countryside Survey integrated assessment

S. Marks, P. Rose, H. Pontier

8.1 Introduction

Chapter 1 provided an overview of the evolving scientific and policy context for the concept of ecosystem services, and identified some of the main challenges involved from a natural sciences perspective. Key scientific questions included how to define and measure complex biological and environmental processes, and how in turn these can be linked to ecosystem services. The integrated assessment also provided insights into drivers of change in the measured ecosystem service, as well as exploring interactions between drivers and separating key drivers and their effects at different geographical and temporal scales. This signalled important messages for policy makers.

This chapter will attempt to show how the key findings for the ecosystem services studied in this report might begin to contribute to the evidence base for policymakers, in particular for applying the concept of ecosystem services to decision-making using the Ecosystem Approach and increasing appreciation of the importance of the effects of cross-cutting policies on biodiversity and the capacity of ecosystems to deliver a range of ecosystem services.

The following discussion is structured around selected policy themes, in contrast to the main report which is organised by ecosystem service.

For each policy theme, some examples of the contribution of this work, and how further work may be useful, will be explored. Some suggestions for how the evidence base might be strengthened in future and general conclusions are also given.

8.2 Ecosystem services and the role of natural sciences

Over the last few years, the Ecosystem Approach (EA, see Box 8.1) and the concept of ecosystem services have become prominent in policy thinking. The demand for new knowledge and synthesis to help understand ecosystem states and changes, and ensure their sustainable use into the future, is apparent through a wide variety of guidance documents, together with benchmarking and economic valuation initiatives operating from global to local scales (see Chapter 1). The main problems have been in quantifying ecosystem services and understanding the role of biodiversity in ecosystem function and supply of services so that policy makers can understand change and responses of ecosystems to anthropogenic pressures and policy interventions at relevant spatial and temporal scales. Many publications on ecosystem services fall into the category of 'grey literature', with relatively little formally published academic research.

The Ecosystem Approach **Box 8.1**

The **Ecosystem Approach (EA)** is a framework for delivering progress towards sustainable development objectives as set out by the Convention on Biological Diversity (CBD), which defines the EA as a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way (JNCC, 2006). It involves consideration of all forms of relevant scientific, indigenous and local knowledge, information and practices, as well as the economic/societal context (CBD, 2007).

So far, there has been a strong policy focus on the challenges of economic valuation of ecosystem services. As discussed in Chapter 1, putting a price on nature is a controversial and difficult undertaking for a variety of reasons and is beyond the scope of this report. However, economic valuation is a priority for policymakers because there is a risk that if nobody pays for services that do not currently have a market value attached to them, decisions may be made which irreversibly compromise future service provision.

Whatever the difficulties of economic valuation, any methods adopted now or in the future will need to be based on a sound scientific understanding of ecosystem processes and functions as well as the synergies and interactions between them. The formulation of possible policy responses to pressures which change ecosystem service delivery will then be based on the best evidence. Thus, the natural sciences have a key role to play in informing the design of robust decision support tools for use when there are conflicting or competing demands on services.

The Ecosystem Approach has been more readily adopted in relation to biodiversity protection in a nature conservation context than in recognition of its supporting role for all ecosystem functioning (and therefore all ecosystem services). It should be emphasised that the EA has relevance for supporting policy thinking about a wide range of policy areas, given that sustainable management and use of all natural resources depends on resilient, healthy and functioning ecosystems.

8.3 Policy relevance of the Countryside Survey integrated assessment

The approach taken for the Countryside Survey (CS) integrated assessment was to explore the potential of the multivariate CS data set containing various measurements of countryside features to quantify and assess changes in a small number of ecosystem services (see Chapter 1; Table 1.2). Although the term '*integrated assessment*' may be defined as an interdisciplinary process for combining, interpreting and communicating knowledge from diverse scientific disciplines (van der Sluijs, 2002), in the context of this report it refers primarily to the integrated analysis of CS soils, water, vegetation and landscape data with other relevant national-scale environmental datasets as a foundation for:

- devising measurements of ecosystem services that can be used to express their current status and changes over time;
- identifying likely causes and drivers of change exploring interrelationships, synergies and trade-offs between different ecosystem services;
- identifying potential implications for a range of policy areas, for some of the services studies.

The CS integrated assessment is integral to the development of the evidence base underpinning the UK National Ecosystem Assessment (NEA)[55], a major initiative under the Living With Environmental Change (LWEC) partnership which aims to assess status and trends in UK ecosystem services. The NEA aims to apply the principles of the Millennium Ecosystem Assessment (MA)[56] to inform and develop policies for long-term sustainable delivery of ecosystem services. This exercise also contributes to the fulfilment of assessment requirements under international and EU legislation such as the CBD and EU Water Framework Directive (WFD). As discussed in Chapter 1, ecosystem services for the integrated assessment were classified using the NEA system (see Table 1.2). At the time of publication, the NEA has produced a preliminary synthesis and progress report on status and trends by Ecosystem Service type and Broad Habitat. The common classification framework between CS and the NEA should facilitate an integrated approach to future synthesis and data analysis.

Several broad themes and their policy context are discussed in this chapter:

- protecting biodiversity;
- climate change mitigation and adaptation;
- freshwater quality;
- agriculture and
- landscape.

However, it is important to recognise that the integrated assessment also showed clearly that biodiversity, ecosystems and service delivery can be affected by interactions between several different policies acting on the same landscape.

[55] http://uknea.unep-wcmc.org/
[56] www.millenniumassessment.org

The potential contribution of evidence from the CS integrated assessment to the policy debate is discussed below.

Protecting biodiversity

Policy context

In line with the revised strategic plan[57] for the Convention on Biological Diversity (CBD) expected to follow on from the 2010 biodiversity targets, the UK Biodiversity Partnership now puts greater emphasis on ecosystem services, rather than focusing solely on species and habitat targets. This change of approach represents a move from the predominantly static, site-based, target-driven nature conservation approaches of previous decades towards more dynamic, process-based approaches that strive to maintain ecological integrity and ecosystem functioning (Bennett *et al.* 2009; Haslett *et al*. 2010), a crucial consideration if biodiversity loss is to be halted beyond 2010 and sustainable delivery of ecosystem services ensured. This will require a more complex science base, which needs time to be developed, but it will result in outputs that are more likely to be based on understanding of the systems involved.

Understanding and defining the role of biodiversity in ecosystems is a complex undertaking. Biodiversity is generally understood as the variety of life on earth which underpins ecosystem structure and function, and hence it supports the delivery of all ecosystem services. The role of biodiversity in ecosystem function and delivery of services is poorly understood. For example, many taxonomic groups are still poorly understood (particularly lower plants and animals, including micro-organisms), in contrast to other taxonomic or functional groups which can be more clearly related to particular ecosystem services (such as nectar-producing plants which can be linked to pollination services).

The chapters of this report use the uniquely fine-grained but large-scale datasets from CS to develop and demonstrate a number of metrics of ecosystem service status and change. The scientific linkage between the biophysical variables and the end-service is described and justified for each relationship using the conceptual ecosystem service cascade of Haines-Young and Potschin (2008). Analysis of current status, trends and possible causes of change in these metrics provides a basis for furthering our understanding of the role of policy in both driving previous change in service quality, mitigating the impacts of previous reductions and planning for multiple service delivery at different scales. Table 8.1 summarises the main roles of biodiversity and policy interactions identified in this study. The table demonstrates the importance of biodiversity in delivery of all these example services, and the complex responses or interactions with a range of different policy areas, and therefore the need to appreciate the impacts of cross cutting policies, as well as geographical and spatial differences in policy for sustainable natural resource management.

[57] http://www.cbd.int/doc/meetings/nr/ws4nrsp-cca-01/official/ws4nrsp-cca-01-sp-prep-02-en.pdf

Table 8.1: Summary of ecosystem services, role of biodiversity in providing those services and policy interactions demonstrated in this report.

Chapter: Service	Role of biodiversity	Policy interactions
C2 :Water Quality (Provisioning)	Biodiversity of habitats in surrounding land and streamsides contribute to water quality, or provisioning ecosystem service.	Land use and management of terrestrial habitat, (especially agricultural land use) and stream bed management affected water quality. Multiple impacts were detected, e.g. arable land use and soluble reactive phosphorous affected water quality as measured by macroinvertebrates. The results would be of interest to rural development, agri-environment schemes, and pollution control and water quality management policy makers.
C3 : 'Appropriate biodiversity' (Cultural)	Habitats are associated with typical species and communities, desirable species are appropriate to each community type, whilst undesirable species indicate degradation of the habitat type and reduced delivery of the cultural ecosystem service.	Climate change and nitrogen deposition affected 'appropriate diversity' (abundance of desirable and undesirable species) for the habitat type in some habitats. Models of effects of climate change and nitrogen deposition on *Sphagnum* spp. demonstrated the potential for model-based assessment of risk to 'appropriate diversity' (upland bogs 2020-2050). The result would interest policy makers for climate change adaptation and mitigation, air quality, and those responsible for maintaining coherent ecological networks under the Habitats Directive.
C4: Soil carbon (Regulating, Provisioning and Supporting)	Biodiversity and habitats influence topsoil carbon through sequestration, litter deposition and decay and influences on water and other nutrient cycling.	While the role of soils in climate regulation through carbon storage remained largely unchanged, habitat cover (fertile grasslands and arable land), and air quality improvements in the form of reduced acid deposition probably affected carbon density on top 15cm of soils, which could have implications for food production and provisioning services.
C5: Pollination (Regulating)	Biodiversity; nectar plants for wild bees which provide pollination services for crops and wild plants. Clear differences in nectar plant diversity are seen between	Climate change and pollutant effects were not detected, but trends towards succession, or lack of management were related to changes in pollinator food plants, suggesting that the results would be of interest for land use and management policy makers. Sheep grazing reduced nectar plants in Dwarf

	Broad Habitats.	Shrub Heath. Maintaining habitat mosaics and small areas of species rich refuges is important in maintaining pollinator food supplies and nectar plant diversity. There may be lag effects in the system following land use changes after the war, when the most rapid habitat conversions and species losses occurred. This has left a landscape with fewer species to lose. Scenarios were used to test possible future policy outcomes, and quantify uncertainties. Changing some habitats e.g. Coniferous Woodland to other habitats could give greater increase in nectar plant diversity than transforming other habitats. Of significance for land use planning, different areas of the country deliver different degrees of different services, but these do not always match the areas where human demand occurs.
C6: 'Charismatic landscapes'	Biodiversity, habitat heterogeneity and spatial differences is important in delivery of this cultural service	Combinations and arrangement of features determine how people value landscape, but not all can be influenced by policy (e.g. relief). The integrated assessment was able to quantify and map 'charismatic landscapes', which may be of interest to those policy makers that affect several aspects of land use, including agriculture, forestry, water quality and biodiversity.
C7: Interactions between services	Biodiversity and habitats can deliver more than one service.	Food production and carbon storage (in soils) were at opposite ends of ecosystem service provision. Not all services can be delivered equally in all parts of the country, due to the soil and climate acting as controlling factors on ecological processes. The landscape can be considered as providing intercorrelated bundles of services, so land management options need to consider priorities and possible trade-offs. Landscape with diverse habitats offer greatest potential for combined ecosystem service delivery. Designation and conservation management could provide multiple services, e.g. water quality, topsoil carbon storage, high biodiversity (but low productivity).

Key points of interest to policy makers emerge from examination of Table 8.1, which involve further development of the science of integrated assessment:

- Biodiversity plays a role in delivery of ecosystem services, and also a means to measure and quantify those services, which is not necessarily related to economic value, but can be used to assess the amount of resource and change.
- The ecosystem approach and integrated assessment can be used to help recognise interactions between ecosystems, delivery of services and the impacts of cross cutting policies.
- There is a need for interdisciplinary working amongst scientists and policy makers, to provide the evidence base that enables the evaluation of possible policy options for sustainable management of natural resources and maintenance of the capacity of ecosystem to deliver services.
- To achieve sustainable land use and management, there is a need to make informed decisions and trade-offs, which could be facilitated by developing models to predict possible future impacts of policy interventions.
- There are ecosystem service surpluses and deficits in different parts of the country, and there is a need to increase understanding of the relationships with differences in human population and service demand and delivery in different parts of the country and expected changes in these patterns of demand in the future.
- There is a need to increase understanding of long-term effects, and time lags in responses of ecosystems to pressures, such as intensification of land use after WWII and the impacts, including spatial and temporal variation on biodiversity and delivery of ecosystem services, which we are still seeing now.
- Although this integrated assessment has enabled quantification of some ecosystem services, it has not addressed economic valuation, which could be taken further in the future to increase our appreciation and management of our natural capital.
- Biodiversity and its role in delivery of ecosystem services should not overshadow the intrinsic value of biodiversity and ecosystems. Development and application of the concepts of appropriate diversity and charismatic landscapes help provide quantitative tools for measuring these cultural, spiritual and aesthetic ecosystem services.

Biodiversity as a cultural ecosystem service

Biodiversity can also be understood in the cultural context of nature conservation, where particular species or groups can be viewed as ‘desirable’ or ‘undesirable’, reflected by their treatment within legislative frameworks and strategies aimed at protecting or managing valued habitats and the species which depend on them. Understanding biodiversity in this way acknowledges that many plant and animal species are appreciated for their own sake, but introduces an element of subjectivity and value judgement, as different groups and species are regarded as desirable or undesirable in different contexts. However, where there is an agreed legislative framework, such as Common Standards Monitoring to assess condition of habitats (with accompanying criteria on desirability), this provides a useful starting point for measuring aspects of biodiversity as a cultural ecosystem service.

Contribution of Countryside Survey integrated assessment

Within the CS integrated assessment, a new term, '*appropriate diversity'*, has been introduced to define biodiversity as a cultural ecosystem service. Preliminary attempts were made to quantify one aspect of this service, by exploring CS plant abundance data for those species which are used as indicators for assessing the condition of UK Priority Habitats (UK Common Standards Monitoring (CSM) Guidance; see Chapter 4). These results provide one possible measure of how nature conservation value of habitats varies from place to place across the UK.

In all CS Broad Habitats analysed (except Arable and Horticulture), there was a decline in the provision of appropriate diversity between 1998 and 2007, which is consistent with the general decline in mean plant species diversity observed in CS as a whole during this period (see Chapter 4). This accompanies a pattern of increased succession towards more competitive plant species that are associated with wetter or shadier conditions, suggesting reduced disturbance or management (Morecroft *et al.* 2009; Carey *et al.* 2008).

The possibility of mapping appropriate diversity using CS data for CSM Indicator species was explored, but several methodological limitations were identified. For example, many CSM species are those which are more likely to be found within designated areas. These are likely to be under-represented in CS sample squares because CS is an unbiased sample of the countryside, with designated sites only incorporated where they happen to fall within sample squares and where site interest features are not explicitly targeted. Also, it is problematic to extrapolate data from the 591 CS sample squares whilst maintaining a meaningful degree of spatial sensitivity, as the expected presence and abundance of species that make up appropriate diversity will vary geographically, so use of an absolute or averaged CSM Indicator list would not be appropriate. Future possibilities for developing suitable models to allow mapping of appropriate diversity are discussed in Chapter 4 and Appendix 4.2.

Relationships between appropriate diversity and possible drivers of change were also explored, but these were not always consistent with expectation or alternative plausible mechanisms. Drivers which did relate to appropriate diversity as expected included climate warming since 1980 and reduced nitrogen deposition (see Chapter 4). Further refinement of methods and improved availability of data would be needed to make a fuller exploration of drivers of change for appropriate diversity if findings are to be useful for informing future biodiversity or agri-environment policies. In particular, high-resolution data on location, history and details of agri-environment management interventions with positive environmental effects are required to balance the relative abundance of data available on management practices associated with degraded conditions.

Future work

Although appropriate diversity and the Site Condition Monitoring process for designated sites both make use of CSM Indicators, it is important to emphasise that these measures are not comparable due to differences in approach. Although further work is needed, the 'wider countryside' approach of CS is a potentially useful complement to future biodiversity strategies, as it is in keeping with a shift towards designing policies that aim to view the landscape as a whole, rather than focusing mainly on protecting species and habitats via designated areas.

Freshwater quality

Policy context

The EU Water Framework Directive aims to establish an integrated approach to the use and sustainable management of freshwater resources and ecosystems. Although the introduction of a system for river basin management planning should promote integration across sectors, there is great potential for applying the Ecosystem Approach. In particular, it could assist in the valuation of ecosystem services which support the provision of clean water. Chapter 2 begins to develop the scientific groundwork which could eventually be useful for informing the design of decision support tools that can incorporate measures of ecosystem service provision and the effects on water quality in relation to biodiversity including the complex interactions between terrestrial habitats and freshwater ecosystems.

Contribution of Countryside Survey integrated assessment

Chapter 2 focused on the ways in which land use and habitat structure affect freshwater invertebrate diversity, which was taken as a measure of water quality (a provisioning ecosystem service). This report discussed existing evidence for links between intensive land use and decreased stream ecological quality, and provided evidence of areas where trade-offs are likely to be an issue. For example, streamside woody cover was found to be beneficial for invertebrates, but not for streamside and in-stream plant diversity, nectar plant diversity and appropriate diversity due to increased shading.

An important aspect of an integrated ecosystem approach is to be able to consider ecosystem services at a variety of scales in the context of multiple land uses. Many previous studies of freshwater ecosystems have tended to focus on large-scale land use data, which has meant that more local effects (such as physical properties of the stream channel) cannot be reliably detected. This study usefully demonstrates that data can be considered simultaneously at multiple scales to show how large-scale and local factors vary in their relative importance in different contexts. For example, evidence of spatial gradients was clear between potential anthropogenic stressors (intensive land-use) and indicators of stream biological quality (streamside woody cover alongside headwater streams).

Future work

Further research which considers the effects of scale and interaction between factors is needed to strengthen the evidence base for how land use practices relate to stream water quality. In particular, this type of work offers potential for informing the design of integrated policies (*e.g.* under CAP reform or the Water Framework Directive) that operate effectively at different scales, from regional to catchment and farm level.

Climate change mitigation and adaptation

Policy context

The Kyoto Protocol came into force on 16 February 2005, with the UK having previously ratified the Protocol in 2002. The UK Climate Change Act (2008) was introduced as a legally binding long-term framework for reducing Greenhouse Gas (GHG) emissions, and sets a target of an 80% reduction by 2050 (based on 1990 levels). In response to these commitments, in March 2010, Defra published the UK Government's Climate Change Plan, which sets out measures for climate change mitigation and adaptation within a sustainable development framework (Defra, 2010). The Devolved Administrations have similar legislative drivers, including the Climate Change (Scotland) Act 2009. Reducing carbon emissions from agriculture and other rural land uses is a major priority for both the UK Government and Devolved Administrations,, whether via improved government, industry and public sector collaboration, or via policy levers and legislation.

The Kyoto Protocol highlights the need to protect and enhance carbon sinks where possible. In order to meet emissions reduction targets, a comprehensive approach to carbon in rural land use will be needed, which will require identifying the potential role of soil carbon sequestration in contributing to government targets to reduce greenhouse gas emissions. Soil strategy, like agriculture, is a devolved matter within the UK, but in the Soil Strategy for England[58] (Defra, 2009a), Scottish Soils Framework (Scottish Government, 2009a) and the Welsh Soils Action Plan consultation,[59] the importance of identifying the potential role of soil carbon sequestration in contributing to GHG emission reduction targets is recognised. Peaty organic soils are important as a store of carbon, particularly in Scotland, and to some extent, Wales. Protection and enhancement of soil organic matter and reduction of GHG emissions from soils are highlighted in the Scottish Soil Framework as two key outcomes and priorities for action.

Contribution of Countryside Survey integrated assessment

Carbon storage in soils is a crucial ecosystem service, but there is a lack of fine temporal resolution data in this area. CS data are more suited for providing a longer-term picture, but there is a need to make rapid progress in addressing the move to a lower carbon economy, based on the best available information. Meanwhile, the need for data that can detect short-term effects of changes in land use policy and practice is being addressed at UK and devolved levels, through research programmes such as the Rural Economy and Land Use Programme (RELU) [60] and the Scotland Rural Land Use Study[61], but it will be some years before such data become available.

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[59] Consultation on the Welsh Soils Action Plan: http://new.wales.gov.uk/consultations/environmentandcountryside/130308welshsoilsactionplan/?lang=en

[60] http://www.relu.ac.uk/about/

[61] http://www.scotland.gov.uk/Topics/farmingrural/Rural/rural-land/land-use-study

In Chapter 3, CS data are used to explore the potential for developing a measure of the ecosystem service of soil carbon storage. The lack of evidence for any large-scale changes in topsoil carbon concentrations suggests that there has been no change in its role in climate regulation. However, decreases in carbon concentration (10-13%) and density (5-11%) in arable topsoils (0-15 cm) were observed, which are likely to have implications for the sustainability of food production.

Future work

A series of GB soil monitoring programmes have reported on soil carbon and organic matter content, at country or habitat level (see Chapter 3). Future work includes the validation of models using understanding of past change to inform decision-making for carbon management for different functions. Further research is also required to reduce uncertainty associated with the net carbon (and other GHG) emissions and sinks which result from different land uses. This would help to inform policy decisions which aim to reduce emissions from land use practices, thereby contributing to meeting UK government and devolved administration targets for reducing carbon emissions.

Agriculture – pollination

Policy context

As part of the next round of reform of the Common Agricultural Policy (due 2013), protection and enhancement of the rural environment is a priority, as is the consideration of the strategic importance of climate change when developing rural and agricultural policy. These priorities should promote a more integrated approach to developing resilience and sustainable management across sectors, whilst the framework of the Ecosystem Approach, supports the exploration of how biodiversity supports crucial processes for agriculture. One such process that is the subject of increasing current attention is that of pollination, which has been identified by the CBD as a key ecosystem function that is threatened globally, reflected by the establishment in 2002 of the International Initiative for the Conservation and Sustainable Use of Pollinators[62]. The service of pollination not only ensures production value in crops, but is critical to the survival and maintenance of the diversity of plant populations (Potts *et al*. 2010). There is evidence that invertebrate pollinators play a significant role in food crop production, but they are declining in numbers, possibly due to habitat loss, climate change and disease or the effects of pesticides (Beismeijer *et al.* 2006; Ricketts *et al.* 2008).

There are several government initiatives aimed at improving honey bee health and minimising disease risks e.g. Healthy Bees, a ten year strategy operating in England and Wales to 2011 (Defra, 2009b) and in Scotland, the Honey Bee Health Strategy (Scottish Government, 2008). However, it is also important to assess the state of, and threats to, pollination services provided 'free of charge' by wild pollinators (e.g. bumblebees), such as habitat loss and climate change. Whilst there is scope to

[62] http://www.cbd.int/agro/pollinator.shtml

address some of these threats within existing biodiversity and agri-environment policies (e.g. measures in England's Environmental Stewardship scheme), there is a need for development of further measures to benefit pollinators, underpinned by sound scientific research. Such research includes the LWEC Insect Pollinators Initiative[63], a five year, £10m research programme which from 2010 will look at both disease and environmental threats to a range of insect pollinators, including both wild and managed species. ALARM (Assessing LArge-scale Risks to biodiversity with tested Methods)[64] is an EU Framework 6 Integrated Project, a major theme of which was the study of risks arising from pollinator loss in the context of current and future European land use patterns.

Contribution of Countryside Survey integrated assessment

The CS integrated assessment did not measure the abundance of pollinating insects directly, but was able to provide an analysis of changes in the diversity of nectar producing plants used by bumblebees and solitary bees, as an indicator of one aspect of potential pollination service delivery in British habitats. CS data from three survey periods between 1990 and 2007 were analysed. Where sample sizes allowed, the largest declines were seen in small patches of botanically diverse habitat embedded within larger areas of Broad Habitat. Losses were a particular feature of lowland agricultural habitats (Improved Grassland, Neutral Grassland and Arable and Horticulture), and woodland Broad Habitats (see Chapter 5). Of particular interest for policy was the implication that landscape-scale nectar plant diversity may rely on habitat mosaics, especially the maintenance of small patches of species rich habitat or linear habitats. These locations are vulnerable to surrounding intensive management whilst a delicate balance is required in terms of favourable disturbance; too little and diversity is suppressed under a shading canopy of trees and shrubs. Observed changes in nectar plant diversity were not explained by external drivers such as climate change or nitrogen deposition. It was not possible to attribute observed changes to any effects of agri-environment scheme management because adequate explanatory data could not be obtained. Attribution results did, however, show that a strong negative relationship between number of nectar plants and sheep numbers in upland heath but this relationship was already apparent by 1990 and did not change up to 2007. Across a number of Broad Habitats, patterns of change in the vegetation suggested that succession (related to reduced management) was associated with loss of nectar plant diversity in all situations except Boundary and Linear Features at the Broad Habitat scale.

Despite the lack of finely resolved data, it was possible to construct models of spatial variation in diversity of nectar-producing plants at the GB Broad Habitat scale, and to produce predictive maps to test of the impact of agri-environment scenarios. This raised interesting questions such as whether the geographical distribution of nectar plants and associated pollinators that might be achieved in response to the policy intervention matched the areas where their services might have been required, such as areas where flowering crops are grown (e.g. rapeseed or sunflowers). The study

[63] http://www.lwec.org.uk/activities/insect-pollinators-initiative
[64] http://www.alarmproject.net/alarm/

demonstrated much potential for exploring multiple impacts of human activities at the fine scale across British habitats.

Future work

The results of this preliminary assessment and modelling work may eventually be useful for informing the design of agri-environment scheme options and other measures aimed at enhancing both pollination services and biodiversity in general. However, further integrated assessment of the status and trends of pollination services at a range of spatial scales, making use of data with appropriate spatial and temporal resolutions is required. The incorporation into the UK NEA of CS integrated assessment outputs, as well as those from other initiatives such as ALARM, should serve as a useful foundation for future development of this work in this important area.

Protecting and managing landscapes

Policy context

Following ratification, the European Landscape Convention (ELC) came into effect in the UK in March 2007. Although not a legally binding EU Directive, the ELC provides a useful framework for protecting, planning and managing landscapes at European level. The UK and devolved administrations and relevant agencies have undertaken to uphold the requirements of the Convention, which include protecting and managing cultural and historic aspects of landscape within existing legislative and administrative frameworks.

Contribution of Countryside Survey integrated assessment

In Chapter 6, preliminary investigations were made into how quantitative CS data could be used with other data and qualitative information data to provide a measure of landscape quality. Two measures of landscape quality were explored, the first provided a measure or overall score based on peoples' perceptions and value of landscape features and the second provided a measure of ecological complexity of the landscape.

The landscape quality measure based on ecological complexity used habitat diversity, total species richness per 1km square and mean species richness per plot for each square, which gave contrasting maps. It did not include a measure of peoples' perceptions or values, but it showed where biologically diverse areas occurred. The second measure assessed landscape quality measure using qualitative expressions of preferences of particular landscape features gathered from a study performed in England. Landscape elements that were strongly associated with high aesthetic value and landscape enjoyment were the area of woodland, water, high ground and coast. When allocated preference scores were applied to each habitat feature, the overall charismatic landscape score for each National Character Area was mapped. This demonstrated the potential for integration with both qualitative social research and other landscape classification system. The components were similar landscape classification systems are used in Scotland and Wales so it would be possible to apply this type of approach at a UK as well as country scale. There is also potential to create maps of these services by extrapolating the 1 km square data across GB using the Land Class stratification that underpins the CS sampling design. The maps produced by both methods demonstrated the great differences between parts of the country where 'cultural services' were provided.

Future work

The inclusion of cultural information in ecosystem services presents many challenges, such as the difficulties of measuring it and how to develop consistent approaches. However, this work has shown that CS data has future potential for contributing to the evidence base for understanding relationships between landscape variables and appreciation of the landscape, and a means of measuring provision of cultural services associated with landscape.

8.4 Discussion

The integrated assessment of data from CS with data from other sources has aimed to improve understanding of the relationships between biodiversity, ecosystem functions and delivery of services, including synergies and trade-offs. As CS is a long-term dataset, it provided a valuable opportunity to assess the status and change over time of the chosen ecosystem services, and by analysing the data alongside other datasets relating to possible drivers of change, allowed exploration of relationships between the quantified service and explanatory variables. Quantitative information about ecosystem services is of most value to policymakers and land managers as it facilitates specific, rather than vague, guidance to ensure ecosystem sustainability (Harrison *et al*. 2010). Where links between an identifiable element of biodiversity and an ecosystem service are obvious, this can be used to inform appropriate environmental management policy and practice, including identifying priorities for cross-sectoral integration. For example, the challenges ahead for the rural sector in achieving carbon emissions reduction targets while working within the global context of increasing demand for food have been clearly identified in government climate change plans.

Analyses of interactions between ecosystem service indicators in Chapter 7 demonstrated that food production and soil carbon storage cannot both achieve maximum values within the same 1 km square scale because the ecological conditions that optimise each service are situated at either ends of a primary gradient of soil and climate across Britain. Characterising these basic constraints on landscape delivery of mixed ecosystem services is an important step towards providing the quantitative evidence needed to inform decision-making about trade-offs between soil carbon storage and food production. This technique also offers potential for detecting relationships that may be less obvious, and indeed, may lead to identification of synergies and trade-offs that challenge commonly held beliefs (Rouquette, 2010). For example, from preliminary analyses of headwater streams within Sites of Special Scientific Interest (SSSIs), management for conservation is not obviously beneficial for plant diversity, but is associated with high diversity soil invertebrates (Chapter 2).

Overall this report has begun to demonstrate how indicators of ecosystem services can be quantified and predicted in terms of the proportion of Broad Habitats present in a 1km square. This also offers the prospect of model-based mapping of ecosystem service indicators across all 1km squares in Britain using the new Land Cover Map for 2007 in combination with the ordination models initially developed in this project.

Some limitations associated with the use of CS data for this type of work were also identified (see Chapter 1). Consideration of these may prove useful for informing the future direction of CS itself, as well as decisions about what nationally important long-term datasets should be collected and managed in the future.

An issue which is of particular importance for development of an integrated research and evidence base is the need for common approaches across all disciplines to defining, measuring, analysing, modelling and valuing ecosystem services. As discussed in Chapter 1, the strength of developing and testing common approaches based on CS data is that it is a long-term dataset incorporating measurements of soil, water, vegetation and landscapes from the same locations.

However, it should be borne in mind that where the properties of the CS or other available data have restricted the choice of methods available for defining and measuring services (see Chapter 1), the methods selected may not be the only or indeed optimum choice for defining and measuring that service. Hence, the methods used, and therefore any results, should be regarded as innovative examples of what is possible, rather than definitive.

Additional analyses based on alternative datasets (existing data, or data planned for future collection and tailored more to specific ecosystem services-related questions) will be required to add to the body of knowledge and evidence base for defining ecosystem services. The Scottish Government's 2011-16 Strategic Research programme for rural affairs and environment[65] has been designed with this in mind, and recognises the need to go beyond demonstration projects and case studies towards unified approaches that can deliver decision-support tools suitable for use at a range of scales (from farm management to national strategic levels).

It is important to emphasise that the focus in this report is on addressing questions from a natural sciences perspective. If fully integrated approaches to understanding ecosystem services are to be achieved as a foundation for informing decision makers, there is a need for collaboration with researchers from economics and social science disciplines to develop robust, consistent and joined-up methods of data analysis, modelling and interpretation of results.

[65] http://www.scotland.gov.uk/Topics/Research/About/EBAR/StrategicResearch/future-research-strategy/Themes/ThemesIntro

8.5 Future work

Data collection and management

If integrated approaches to understanding ecosystem services are to progress and produce meaningful analyses incorporating attribution of change at a range of spatial and temporal scales, data on explanatory variables are required at a sufficiently high resolution. The attribution of ecosystem services to possible explanatory variables within the CS integrated assessment was to an extent hampered by a lack of coordinated, spatially coherent data on land management impacts. It is also important that any analyses incorporate data that are sufficiently comprehensive to ensure that the resulting evidence on drivers of recent and future countryside change is robust and balanced. In particular, data on the location, history and detail of management impacts funded under agri-environment schemes were lacking. This risks biasing attribution analyses toward changes driven by negative drivers simply because explanatory variables for these drivers are more readily available.

There is also a need to address issues of data availability and compatibility, as many of the required datasets are not consistent between UK countries. This will require a more strategic approach to future data collection and mechanisms for accessing it across disciplines and sectors.

Interdisciplinary research

Natural and social scientists and economists need to develop common approaches to data analysis and modelling, as at present biophysical data is difficult to interpret and apply across disciplines. Common methods need to be developed for measuring (and valuing) ecosystem services to allow more reliable data interpretation and analysis and more reliable economic modelling. This is essential if we are to manage our natural resources fairly, sustainably and strategically. This will require a high level of integration across natural and social sciences, with input from economists, policymakers and other relevant stakeholders.

Communication and Knowledge Exchange

There is still a need to close the gap between the differing aims and perceptions of researchers and policymakers with regard to ecosystem service research including assessment initiatives such as this project. These research agendas yield important evidence for policymakers in need of more holistic guidance on opportunities and constraints on delivery of multiple benefits at multiple scales, but more robust evidence is still required to inform decisions where there are competing priorities or multiple demands on resources. In the research environment, the emphasis is on working towards a better understanding of ecosystem processes and how the drivers of change operate at different spatial and temporal scales. Such knowledge should strengthen environmental decision-making over the long-term, whilst allowing for the evolution of policymaking frameworks and any new approaches that may be adopted in the future.

8.6 Conclusions

Development of a scientific evidence base is required to inform decision-making using the concept of ecosystem services, but it is still in its early stages. The integrated perspective across ecosystems and development and testing of novel methods paves the way for further research, particularly for exploring interrelationships between ecosystem services and the factors which drive change over time. However, as demonstrated by this assessment, it can sometimes be difficult to attribute patterns in the results to potential driving factors, and relationships detected for some variables may be unexpected or inconsistent. This demonstrates the complexities involved in measuring ecosystem services and the challenges of measuring how service provision varies geographically and through time.

The scientific focus therefore tends to be on continuing to explore novel methodologies. In many cases further data collation, analysis and interpretation will be required before accessible information suitable for evidence-based policy making becomes available. Given that the Ecosystem Approach is already being adopted by decision-makers at all levels, this offers a challenge to economists and scientists from all disciplines to work together with decision makers and other stakeholders to meet the rapidly growing demand for an integrated evidence base.

Integrated assessment can be viewed as a positive way forward for dealing with complex environmental problems which cut across the boundaries between disciplines. The novel scientific work presented here should serve as a useful basis for strengthening the policy evidence base in future years.

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Glossary

The following is a table of terms which will be used in this report, together with their meanings as used here. It is recognised that semantic issues can dominate definitional issues in relation to ecosystem services, hence the need to be clear how the various terms below are used in this report. In order to be as consistent as possible with the widespread use of terms, this glossary has been cross-checked with that of the Millenium Ecosystem Assessment (MA 2003), work under RUBICODE (Harrison et al, under review) and the recent paper on the classification of ecosystem services for valuation by Fischer & Turner (2008).

Term	Definition
Attribution	The process of identifying variables which have a causal effect on a given parameter.
Bayesian probability	A subjective characterisation of probabilities of outcomes arising from a certain decision.
Biodiversity	The variability among living organisms from all sources including terrestrial and freshwater ecosystems and the ecological complexes of which they are part; this includes diversity within and among species and diversity within and among ecosystems.
Biomass	The mass of living tissue in either an individual or cumulatively across organisms in a population or ecosystem.
Broad and Priority Habitats	A classification of UK habitats produced for UK Biodiversity Action Plan reporting.
Cultural services	The nonmaterial benefits that people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation and aesthetic experience, including, for example, knowledge systems, social relations, and aesthetic values. These may also be seen as 'cultural benefits' as they directly relate to changes in human welfare.
Driver	The underlying causes of change in an ecosystem which may be human induced or natural.
Ecosystem	A dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit.
Ecosystem benefits	A benefit is something that has an explicit impact on changes in human welfare, e.g. improved walking conditions or decreased flooding.
Ecosystem services	A collective term to describe ecosystem functions and processes which have human beneficiaries. These include a range of intermediate services which may be involved in **regulation** (e.g. flood control), **support** (e.g. nutrient cycling) or **provisioning** (e.g.

	pollination) of ecosystems. These services are essential for maintaining conditions for life on earth. For valuation purposes it may be preferable to consider **provisioning** services like food production or **cultural** services such as spiritual and recreational as Ecosystem benefits (Fischer & Turner 2008).
Ecosystem function	An intrinsic ecosystem characteristic related to the set of conditions and processes whereby an ecosystem maintains its integrity (such as primary productivity, food chain, biogeochemical cycles). Ecosystem functions include such processes as decomposition, production, nutrient cycling and fluxes of nutrients and energy.
Final services	These services derive from a range of intermediate services and result in a direct benefit to humans e.g. provision of clean water.
Geographic Information System (GIS)	A computerised system organising datasets through a geographic referencing of all data included in its collections. A GIS allows the spatial display and analysis of information.
Habitat	Area occupied and supporting living organisms. Also used to mean the environmental attributes required by a particular species or its ecological niche.
Indicator	A simple measurable and quantifiable characteristic responding in a known and communicable way to a changing environmental condition, to a changing ecological process or function, or to a changing element of biodiversity. In this report official indicators such as species used in Common Standards Monitoring are denoted by a capital 'I'.
Interactions including trade-offs	In all ecosystems interactions between different ecosystem services may occur. In some cases different services may be positively related with one another, and in others the reverse may occur, e.g. a decrease in the nutrient cycling capacity of soil as a result of its use for food production under particular agricultural systems. The latter situation may be referred to as a trade-off between services.
Intermediate services	A process or measure which contributes to a final ecosystem service but does not, of itself, constitute that service, these include **supporting, regulating** and some **provisioning services**.
Land cover	The physical coverage of land usually expressed in terms of vegetation cover or lack of it. Influenced by but not synonymous with *land use*.
Land use	The human utilisation of a piece of land for a certain purpose (such as agriculture or recreation).

Landscape	An area of land that contains a mosaic of ecosystems, including human-dominated ecosystems. The term *cultural landscape* is often used when referring to landscapes containing significant human populations.
Less/more productive species	Plant species differ in their inherent ability to exploit resources and accumulate biomass.Where resources are not limiting (eg. soil nutrients, light and water) productive species grow more quickly and can outcompete smaller, less productive species inherently less able to exploit high levels of resources. More productive species include wheat, stinging nettle and perennial rye-grass. Less productive species include wild thyme, bilberry and butterwort.
Less/more productive assemblages	More productive species assemblages reflect fertile conditions. In the extreme they are often species poor because more productive species are able to outcompete many other plants. At the other extreme, assemblages of less productive plants may also be species poor per unit area because plants are small in size and mortality is high. The highest species richness is often realised in assemblages that reflect intermediate productivity.
MA	The Millennium Ecosystem Assessment
Model	Mathematical approaches which attempt to describe real world relationships between a range of parameters in order to further understanding of ecosystems and enable prediction of future conditions under different scenarios.
Natural Capital	The stock of biodiversity contained within a particular habitat or ecosystem from which all ecosystem services are derived.
NEA	The National Ecosystem Assessment for the UK
Pollination	The completion of the sexual phase of reproduction in some plants by the transfer of pollen. In the context of ecosystem services, pollination generally refers to animal-assisted pollination, such as that done by bees, rather than wind pollination.
Primary production	Assimilation (gross) or accumulation (net) of energy and nutrients by green plants and by organisms that use inorganic compounds as food.
Projection	A potential future evolution of a quantity or set of quantities, often computed with the aid of a model. Projections are distinguished from ‘predictions’ in order to emphasis that projections involve assumptions concerning, for example, future socioeconomic and technological developments that may or may not be realised; they are therefore subject to substantial uncertainty.

Provisioning services	The products obtained from ecosystems, including, for example, genetic resources, food and fibre and fresh water. The end products may be seen as ecosystem benefits.
Regulating services	Intermediate services which involve the regulation of ecosystem processes, including, for example, the regulation of climate, water, and some human diseases.
Resilience	The capacity of a system to tolerate impacts of drivers without irreversible change in its outputs or structure.
Scenario	A plausible and often simplified description of how the future may develop, based on a coherent and internally consistent set of assumptions about key driving forces (e.g. rate of technology change, prices) and relationships. Scenarios are neither predictions nor projections and sometimes may be based on a 'narrative storyline'. Scenarios may be derived from projections but are often based on additional information from other sources.
Supporting services	Intermediate ecosystem services that are necessary for the production of all other ecosystem services. Some examples include biomass production, production of atmospheric oxygen, soil formation and retention, nutrient cycling, water cycling and provisioning of habitat.
Taxa	Nested groups of species that reflect similarity. Familiar taxa are birds (which belong to the class *Aves*).
Uncertainty of maps and models	It is possible to use mathematical models to plot maps of both current and future distributions of a range of variables relating to ecosystems. Inevitably both models and maps are subject to statistical variability due to error in measurement, sampling or variation in the measured variables. A measure of 'uncertainty' may be provided with maps and models to demonstrate the level of potential error associated with them.
Upscaling	The process of aggregating or extrapolating information collected at a fine resolution to a courser resolution or greater extent.

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For further information on Countryside Survey see www.countrysidesurvey.org.uk

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Countryside Survey

NATURAL ENVIRONMENT RESEARCH COUNCIL

Centre for Ecology & Hydrology
NATURAL ENVIRONMENT RESEARCH COUNCIL

defra
Department for Environment Food and Rural Affairs

NATURAL ENGLAND

Llywodraeth Cynulliad Cymru
Welsh Assembly Government

The Scottish Government

NIEA Northern Ireland Environment Agency
www.ni-environment.gov.uk

An Agency within the Department of the Environment
www.doeni.gov.uk

Forestry Commission

Cyngor Cefn Gwlad Cymru
Countryside Council for Wales

Scottish Natural Heritage
All of nature for all of Scotland

Further information on Countryside Survey can be found at:

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