



Review

Land use and biodiversity relationships[☆]

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ABSTRACT

The relationships between land use and biodiversity are fundamental to understanding the links between people and their environment. Biodiversity can be measured in many ways. The concept covers not only the overall richness of species present in a particular area but also the diversity of genotypes, functional groups, communities, habitats and ecosystems there. As a result, the relationships between biodiversity in its broadest sense and land use can be complex and highly context dependent. Moreover, the relationships between them are often two-way, so that simple relationships between cause and effect can be difficult to identify. In some places, specific land uses or land management practices may be important in sustaining particular patterns of biodiversity. Elsewhere, the uses to which land can be put are highly dependent on the biodiversity resources present.

The review will consider how changes in the quantity, quality and spatial configuration of different aspects of land use can impact on different components of biodiversity, and what direct and indirect factors might drive these changes. The need to distinguish between land cover and land use will be discussed in relation to the economic and social drivers of land use change. The review will also consider whether framing biodiversity objectives involves society in placing constraints upon the types of land use and management practice that are possible, and will consider such arguments in relation to assessments of the costs of biodiversity loss. It would seem that while considerable progress has been made in mapping out plausible futures for land use and biodiversity at global and regional scales, closer integration of modelling, scenario and field-based monitoring is needed to strengthen the evidence base available to decision makers. Challenges that face us include how we take account of the qualitative changes in land cover, and the impacts of such modifications on biodiversity and ecosystem services. Broader perspectives on the value of biodiversity and ecosystem services are also needed as the basis for developing adaptive and flexible approaches to policy and management.

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Contents

Introduction	S178
Current understandings	S179
Data infrastructures	S180
Trends in land use and biodiversity	S181
Key scientific challenges	S183
Evolving policy perspectives	S184
Conclusions	S184
References	S185

Introduction

An awareness of the relationships between land use and biodiversity is fundamental to understanding the links between

people and their environment. On the one hand, land use change and transformations in the way land is managed are key drivers of changes in biodiversity at global, national and local scales. On the other, given the need to sustain ecosystems and the benefits that people derive from them, the biodiversity of a site, or of an area of land, may often place constraints on our choices about how it can be used. So important is the topic that Turner et al. (2007) have argued that 'land change science' has now emerged as a central component of global environmental and sustainability research. By 2100, the impact of land use change on biodiversity is likely

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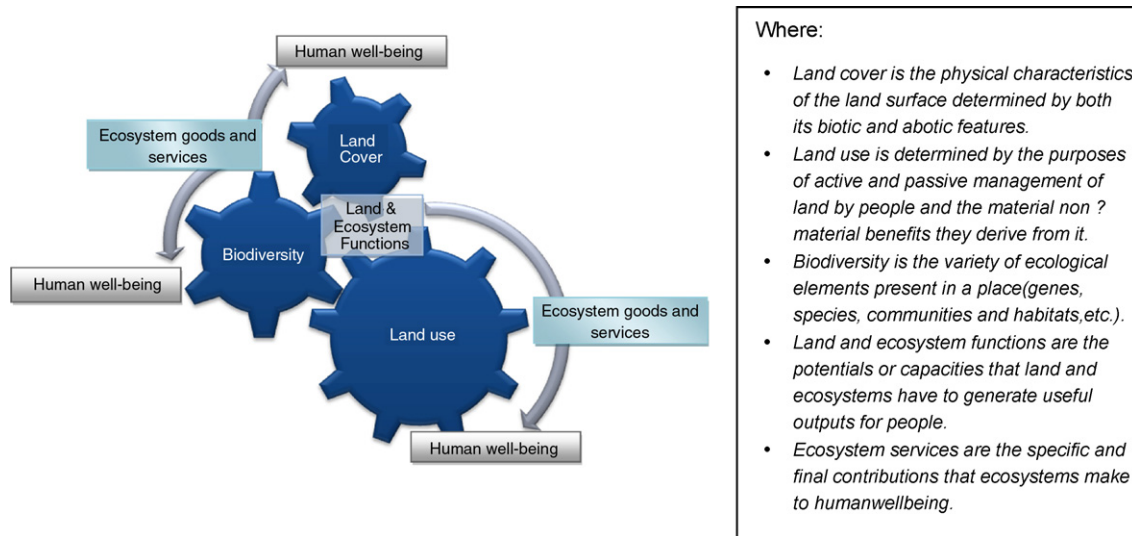


Fig. 1. The relationships between land use, land cover, biodiversity and the output of ecosystem services.

to be more significant than climate change, nitrogen deposition, species introductions and changing atmospheric concentrations of carbon dioxide at global scales (Chapin et al., 2000; Sala et al., 2000).

The aim of this review is to take stock of existing insights into the relationships between land use and biodiversity. It will evaluate the evidence on which understandings of current trends are based, and explore efforts to identify 'possible futures' through scenario and modelling studies. It will conclude with an assessment where key gaps in our knowledge of the relationships between land use and biodiversity are, and how a better understanding of the issues might help shape future management and policy responses. The need to focus on such issues is particularly pressing given, for example, the conclusions of the mid-term review of the EC Biodiversity Action Plan (Commission of the European Communities, 2008). This suggests that the EU commitment to halting the loss of biodiversity by 2010 is unlikely to be met. A similar conclusion has been reached in the UK (House of Commons Environmental Audit Committee, 2008). Questions about the scale and type of land use change that is required to reverse such trends have been identified as one of the key issues of high policy relevance in contemporary ecology, both in the UK and elsewhere (Sutherland et al., 2006, 2009). Vickery et al. (2004), for example, has suggested that if we are to halt the decline of farmland birds in England, the area of 'sympathetically-managed land' needed to reverse current trends is significant.

Current understandings

The study of the relationships between what we term 'land use' and 'biodiversity' is inherently complex because both concepts are multi-faceted and difficult to define unambiguously. Biodiversity can be measured in many ways. The concept covers not only the overall richness of species present in a particular area, but also involves measures of the diversity of genotypes, functional groups, communities, and ecosystems that might be identified (e.g. de Bello et al., 2008). Some commentators have even used the term to cover the diversity of products and services an ecosystem can provide (Mace et al., 2005), the 'multi-functionality' of ecosystems.

It is generally accepted that it is not possible to represent biodiversity by a single indicator, and that a multidimensional approach is needed to understand implications of changes biodiversity for

ecosystem functioning and the ecosystem services (MA, 2005). In this review the term biodiversity will be used in its broadest sense and qualified where necessary to make it clear what aspect is being considered. The same kind of argument applies to the analysis of the relationships between biodiversity and land use. While the two may be linked in very general terms, we have to be specific about what particular components of biodiversity and land are being considered if we are to understand the cause-effect relationships that might exist between them. The task is a challenging one because definitional problems also exist in relation to the notion of 'land use'.

It is widely acknowledged that 'land cover' and 'land use' are not the same thing (Jansen and Di Gregorio, 2002; Comber, 2008). 'Land cover' refers to the physical surface characteristics of land (for example, the vegetation found there or the presence of built structures), while 'land use' describes the economic and social functions of that land. Clearly the two may be linked, but the linkages are complex. A single type of land cover, perhaps grassland, may support many uses, such as livestock production, recreation and turf cutting, while a single use, say mixed farming, may take in a number of different cover types including grassland, cropped and fallow areas. However, while the distinction between cover and use is accepted, they are often conflated in classification schemes (Jansen and Di Gregorio, 2002), so that resulting information on change is difficult to interpret, particularly in terms of its consequences for biodiversity. In the context of understanding the links between land and biodiversity, it is not always clear quite what 'land use change' means. Does it mainly refer to gross changes in which there is complete replacement of one type of cover or use by another, or does it also include the more qualitative changes in the characteristics of land? These latter are what Lambin (1999) has described as 'land cover modifications', and he suggests they are probably more common than wholesale conversions. These kinds of change are subtle and often difficult to characterise, but their implications for the biodiversity characteristics of the land can, as we shall see, be as important as a complete transformation.

Turner et al. (2007) suggest that amongst the challenges facing 'land change science' is the need to develop new and better methods for characterising land. Although they were thinking more generally, this is especially true in the context of biodiversity, given the economic and social arguments that are currently being advanced for conserving, protecting and restoring ecolog-

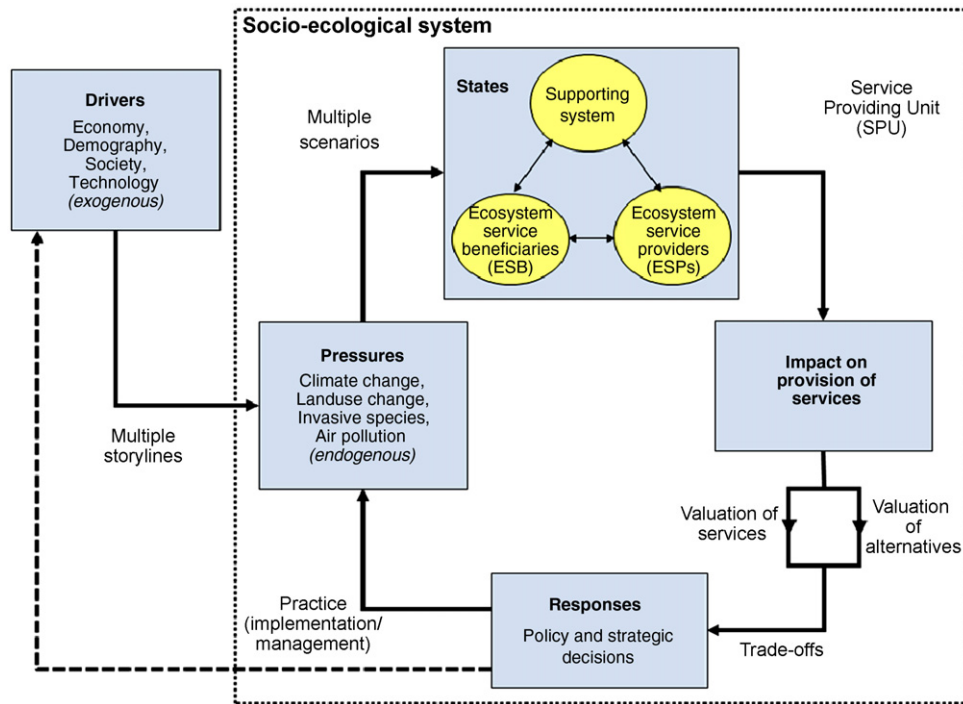


Fig. 2. A framework for linking direct and indirect drivers, pressures and responses in a coupled socio-ecological system for assessment of the effects of environmental change drivers on ecosystem services (after Vandewalle et al., 2008).

ical systems. In recent years, studies of land use and land cover change have become increasingly interdisciplinary (Rindfuss et al., 2008). Nowhere is this more obvious than in the characterisation of ecosystem services. Notions of 'land functions' (Verburg et al., 2009), 'land use functions' (Perez-Soba et al., 2007), and 'landscape function' (Haines-Young, 2000; De Groot, 2006; Kienast et al., *in press*) have emerged as a way of tracing the connections between land and the ecological systems and the ecosystem services that it supports. The term 'function' is used to identify some capacity of land or an ecosystem to generate a service that ultimately provides a benefit to people; by extension the notion of 'multi-functionality' is now widely employed to describe the multiple benefits that land and ecological systems more generally might generate.

Fig. 1 highlights the mutual interdependencies between land cover, use and biodiversity. These are the focus of much current research. It also suggests some broad definitions of the terms as they are used throughout this paper. The various components of biodiversity (at the individual, population and community levels) and the ecological functions that they support have a central place in the emerging understandings of how people and ecosystems are connected, often through the lens of land use and land cover. The physical aspect of land cover depends on, and is influenced by, the uses to which land is put and its biodiversity characteristics. Similarly the range of potential uses that an area of land can support is constrained by and determines the resulting land cover and its ecological status. Much recent work has attempted to better understand this multi-dimensional system.

The idea of a service providing unit (SPU), for example, was proposed by Luck et al. (2003) who argued that while a population or organisms could be defined along geographic, demographic or genetic lines, it could also be delimited by the service or benefit it supports at a given scale. Thus an SPU might comprise all those organisms contributing to the wildlife interest of a site or region, or all those organisms or habitats that have a role in water purification in a catchment. It is a kind of ecological 'foot-print' of the service. Vandewalle et al. (2008) and Luck et al. (2009) have gone on to show

how the idea can be linked into the concept of a social-ecological system, how the different pressures and drivers impact upon it, and the relationships between the 'ecosystem service providers' (ESPs) and the 'ecosystem service beneficiaries' (ESBs).

Models such as those shown in Fig. 2 are providing a rich understanding of the links between biodiversity, ecosystem services and human well-being, and help to identify the kinds of trade-off that might have to be considered between services under different land management or land use strategies. They also support the argument that, at least in the context of biodiversity, land use change science is becoming 'multi-theoretical'. This contrasts with the predominantly empirical or 'atheoretical' approach that has characterised the field in the past (Lambin and Geist, 2006; Geoghegan et al., 1998).

Data infrastructures

Theoretical and conceptual advances have to be supported by better observational and monitoring capabilities if progress is to be made. In recent years there have been considerable advances in earth observation technologies that have enabled both land cover and the biophysical characteristics of the land surface to be measured at global, regional and local scales (Strahler et al., 2006). Global land cover maps have been created using data from the AVHRR (Loveland et al., 2000), SPOT-Vegetation (e.g. GLC2000, see Bartalev et al., 2003; Bartholomé and Belward, 2005), and MODIS (Friedl et al., 2002) sensors. The most recent is the GlobCover initiative (Arino et al., 2007), which has resulted in the production of a global land cover map at 300 m resolution using MERIS data acquired between mid-2005 and mid-2006. It updates other existing comparable global products, such as GLC2000 which has a much coarser spatial resolution of 1 km. An example of land cover mapping at regional scales includes the production of CORINE Land Cover 1990 and 2000 in Europe, using data derived from a range of satellite platforms (EEA, 2006).

Beyond the mapping of land cover, capability now exists to monitor many of the physical and biological characteristics of the land. Net primary productivity can be measured on an annual basis at global and regional scales. Attributes such as the Leaf Area Index and Leaf Chlorophyll Content can be measured at the canopy level (e.g. Foody and Dash, 2007). Remotely sensed data can also provide a range of measures of the structural properties of vegetation canopies, so that the potential impact of human use of the land can be monitored. The analysis of the impacts of habitat fragmentation and the interactions between landscape patterns and ecological processes will also continue to be an important focus of these broad-scale studies (Potschin and Haines-Young, 2006).

In moving from the physical characterisation of land cover through to the identification of land use and its associated ecological and socio-economic functions, observational data has to be supplemented with additional ecological and socio-economic information (Kerr and Ostrovsky, 2003). Cardille and Foley (2003) have shown how remotely sensed land cover data can be fused with agricultural census information to develop maps of agricultural land use in Amazonia. Elsewhere, Paruelo et al. (2001) have combined NDVI measurements with land use data to characterise the impact of human activities on ecological function. NDVI is an indicator derived from multi-spectral remotely sensed data which can be used to assess vegetation vigour. In the future it is likely that there will be greater integration of fundamentally different data sources. There is a clear need to track change in land cover, land use and its functional characteristics. But these data need to be combined with other sources of information to assess the implications for biodiversity and ecosystem services.

The GlobCover project is built on collaboration between the European Space Agency (ESA), the European Environment Agency (EEA), FAO, GOF-C-GOLD, IGBP, the European Commission's Joint Research Centre, and UNEP, and as a result we are seeing much greater standardisation between different datasets. The GlobCover classification is compatible with the UN Land Cover Classification System (LCCS), which aims to be both scale and source independent (Di Gregorio and Jansen, 2000). GlobCover products comparable to the European CORINE Land Cover classification are also planned. In addition to the classification of land cover, and the growing potential to map change over time, the ability to monitor more subtle modifications to the biophysical characteristics of land is likely to be the area where the greatest future advances will occur.

Although remote sensing is potentially an important source of information for land cover and use at broad strategic scales, generally there has been limited integration of such sources with data collected on the ground so that the more subtle aspects of change can be detected. Bunce et al. (2008) have argued that there is a need to develop rigorous sample-based methods that can monitor change in the aspects of land cover and biodiversity that cannot easily be detected from space- or air-borne sensors. Building on the experience of Countryside Survey in the UK, they propose and test a stratified sampling method for recording and monitoring habitats and their associated vegetation characteristics at European scales. They argue that if used operationally, such a sampling framework could provide policy makers with the kinds of empirical or observational evidence they need to monitor the drivers of change in the wider countryside, such as intensification and extensification, and potentially the effects of broad-scale policy interventions such as those associated with agricultural policy and agri-environmental schemes. Such efforts are only likely to be effective if these sample-based approaches are integrated with the broader perspectives that remote sensing can bring, so that a multi-scale monitoring system can be constructed.

Trends in land use and biodiversity

Earth observation data have been used as the basis of a recent review of the most rapid land cover changes on a global scale in the past 20 years (Lepers et al., 2005). The analysis suggested that the fastest changes are occurring in Asia, especially in dryland areas. High rates of tropical deforestation have taken place in the Amazon Basin and South East Asia, where it is associated with the expansion of croplands. High rates of urban expansion are also seen in the tropics. Deforestation as a result of logging activities is apparent in Siberia. In the south east of the United States and eastern China there appears to be a rapid decline in the area of arable land.

A number of studies have gone on to model future trends. Tilman et al. (2001) used a range of univariate and multiple-regression models to predict likely changes in crop and pasture areas, and in pesticide and fertiliser use. These models were based on trends for population and GDP extrapolated to 2020 and 2050. They estimated that the global agricultural area would increase by about 18 per cent between 2000 and 2050, but noted that this was only a net figure. Since there is the possibility of withdrawal from agriculture in some developed countries, the expansion in developing regions is likely to be much higher than the net figure, potentially resulting in the consumption of about half of the suitable land in these areas. The more recent OECD Environmental Outlook 2030 (OECD, 2008) confirms this estimate, suggesting a 10 per cent increase by 2030. This study predicts a substantial expansion of agricultural areas in Africa, South Asia and South East Asia. Much of this would be at the expense of forestland. The OECD study 'base-line scenario' assumes no new policies in response to environmental pressures, or on subsidies to agricultural production or on tariffs in agricultural trade. On this basis, the area of mature forests is likely to reduce by 68 per cent in South Asia, 26 per cent in China, and 24 per cent in Africa over the period to 2030.

The basis of the OECD study was the Integrated Model to Assess the Global Environment (IMAGE), which has been developed to understand the relative importance of major processes and interactions in the society–biosphere–climate system (Bakkes and Bosch, 2008). The particular aspect of the study that is of interest here is the link it makes between change in land cover and land use and biodiversity, most recently explored in the 'Cost of Policy Inaction' analysis undertaken as part of the TEEB initiative (European Communities, 2008; Braat and ten Brink, 2008). Within the IMAGE model, spatial patterns of land use change are calculated from the simulation of regional production of food, animal feed, fodder, grass and timber, information on local climatic and terrain conditions, and changes in natural vegetation due to climate variation. These land use changes, coupled with other pressures also derived from the modelling framework, for climate change, nitrogen deposition, habitat fragmentation, and the expansion of human settlement and developed infrastructure, are then used to simulate impacts on biodiversity. This step is achieved by coupling the IMAGE output to GLOBIO3. The latter models biodiversity impacts through a set of 'dose–response' relationships constructed from a review of the peer-reviewed literature (Alkemade et al., 2006). Both the OECD study and the COPI work provide an indicator of biodiversity loss based on the ratio between the mean species abundance of the original species complement remaining in an area and the mean abundance of species in the natural or low-impacted state.

The COPI study notes that human activities have long influenced biodiversity, and that by 2000 about 73 per cent of pre-human global natural biodiversity was left, with the greatest declines in the temperate and tropical grasslands and forests. Under the OECD baseline scenario it is estimated that between 2000 and 2050, a further 11 per cent loss would occur. The OECD countries show the lowest rates of loss, and here the main pressure is the

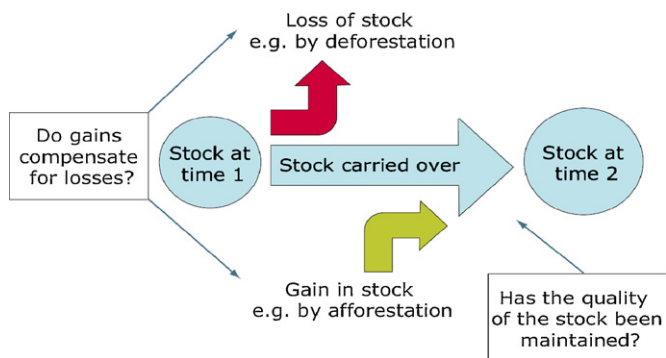


Fig. 3. The basis of land and ecosystem accounts (after EEA, 2006).

expansion of infrastructure. For the so-called BRIC group (Brazil, Russia, India and China) rates of loss are around the global average, and here the main pressure is agricultural expansion. For the 'rest of the world' the effects of climate change coupled with infrastructure and agricultural expansion combine to generate the highest rates of loss.

In contrast to these areas of rapid land cover change, land cover in Europe appears to be comparatively stable. The most recent picture is provided by land cover accounts derived from a comparison of the Corine Land Cover data for 1990 and 2000 (EEA, 2006). These data have much finer resolution than that used for the global studies, and make some hotspots of change apparent. There are notable areas of urban expansion around existing urban centres extending from the southern part of the UK into Belgium, the Netherlands, Denmark, Germany and northern France, and along the Mediterranean coast; much of the land lost to urban use is former agricultural land. Elsewhere, in Spain and Greece, conversion of forest and semi-natural land to agriculture appears to be taking place. In marginal areas, such as the mountain regions of Europe, and in Hungary, Slovakia, Portugal and Italy as well as in some parts of Germany, agricultural abandonment appears to be occurring. However, more than 97 per cent of the land area of Europe retained the same cover in 2000 as was recorded in 1990. Data from successive Countryside Surveys in Great Britain also reveal patterns of stability. Only 6 per cent of the land surface area has changed its cover between 1998 and 2007 (see Carey et al., 2008); a similarly slow rate of change was recorded by in Countryside Survey 2000 (Haines-Young et al., 2000, 2003a,b).

Although rapid land cover change is likely to have major implications for biodiversity, stability of land cover does not mean that there are no impacts or pressures upon ecological systems. As the EEA (2006) land accounting study notes, two issues need to be considered when considering land cover and land in relation to the goal of sustainable development (Fig. 3). The first is whether the gains and losses to the stocks of a particular land cover types are in some sense compensatory. The second is whether the quality of the land stock carried over from one time to another is maintained. In other words, has the capacity of land to generate ecosystem services and support biodiversity been retained over an accounting period?

In the US, Rudel et al. (2005) have explored the compensatory nature of different types of change in land cover and use in the context of the re-establishment of forest following withdrawal of agriculture or active replanting. They found that these 'forest transitions' did little to conserve biodiversity, but may enhance carbon sequestration and soil quality. Unfortunately, few studies of this kind are available to permit generalisations about other kinds of transformations in cover, although the literature on restoration ecology might offer some clues and insights. It would be particularly valuable to attempt to build the notion of compensatory flows into

global models so that the implications of various transformations in land cover and land use can be more finely resolved.

In the UK, the Countryside Survey illustrates one approach to the problem of monitoring change in the more qualitative characteristics of land using indicators derived from vegetation composition. On the basis of their known ecological distribution, plants can be given scores according to their responses to different environmental gradients, such as fertility, moisture and light (Hill et al., 2000). Using these so-called 'Ellenberg scores', Countryside Survey 2000 showed that while the stock of different land covers in the uplands, for example, was hardly changing, their capacity to support the range of plants characteristic of these habitats had declined between 1990 and 1998, due to eutrophication (Haines-Young et al., 2000, 2003a,b). Countryside Survey 2007 showed that between 1998 and 2006 the rate of change had slowed (Carey et al., 2008). The successive surveys also indicated that in woodland habitats indicators based on the sensitivity of plants to changes in light regime suggest the development of more shaded conditions in forest stands. The system of plant indicators and more general vegetation monitoring methods used in Countryside Survey does not make it possible to link the changes in ecological characteristics recorded to particular drivers. However, as Smart et al. (2003) note, the kinds of change observed are consistent with the expected effects of influences which are known to have been important over the latter part of the 20th century in the UK, including increased sheep grazing in the uplands, the increases in nitrogen deposition, agricultural intensification in the post-war period, and the combined effects of under-utilisation and eutrophication of agriculturally marginal habitats and linear landscape features in the lowlands.

When thinking about the implications of land use change for biodiversity at global scales, it is perhaps inevitable that the focus of the analysis has to be on the more easily detected transformations or conversions of one type of cover or use to another. However, the kinds of qualitative change detected by more focused and local studies such as Countryside Survey may also be significant at these much broader scales. Quite different conclusions about the possible significance of future global trends emerge, if the intensity of use and its potential effects are taken into account. Balmford et al. (2005) have considered how sensitive the future requirements for cropland are under different scenarios for increased in agricultural yields. Their analysis of the effects of assumptions about potential future crop yields for the 23 most energetically important crops suggests that the impact of differences in yield on the predicted agricultural area needed in 2050 is as significant as the effects of other drivers such as population size or per capita consumption. This suggests that those looking at the impacts of agricultural conversion on biodiversity at global scales should be as concerned with changes in these qualitative influences on land use as with other drivers such as population and consumption patterns.

At European scales, Reidsma et al. (2006) have likewise argued that biodiversity in agricultural areas depends mainly on the intensity of land use, measured by such factors as the amounts of chemical fertiliser or pesticides applied, and output intensity measured as production per unit area and time. Using farm accountancy data they characterised holdings according to the intensity of operations and used these data to predict impacts on biodiversity. This methodology was similar to the one which underpinned GLOBBIO3, and included comparisons across land use–biodiversity loss gradients, comparisons of the biodiversity impacts of different types of farm, and measures of the biodiversity impact of a shift from conventional to organic farming. Using these relationships, they were able to model a range of plausible futures involving changes in the intensity of agricultural activities across Europe.

Key scientific challenges

A recurring theme of the work reviewed here is the need to combine localised insights about the effects of different land management activities or land use conversions on the various components of biodiversity with broad-scale data or models that allow the wider impacts of land use patterns to be assessed. The development of this interface is one of the most important and immediate scientific challenges facing land change science. One priority is to better understand the relationship between land use changes, primary productivity and species diversity. This may be one way in which the impacts of more subtle and qualitative changes in land characteristics on ecological function might be monitored.

Haberl et al. (2007) and others (e.g. Imhoff et al., 2004) have argued that an important indicator of the impact of human activity on biodiversity and ecosystem function is the 'human appropriation of net primary production (NPP)' or HANPP. They estimate that people appropriate more than a quarter of that available and that of this more than half is consumed through harvest. The rest is made up of land induced productivity changes (40 per cent of this quarter) and losses through human induced fires (7 per cent). For Europe, Imhoff et al. (2004) estimate that the levels of human appropriation may be as high as 72 per cent. Haberl et al. (2007) regard such large-scale appropriation at global scales by just one species as 'remarkable', and conclude that it is now having major impacts on the earth's biogeochemical cycles and on the ability of ecosystems to deliver services critical to human well-being.

Firbank et al. (2008) suggest that HANPP is potentially the ideal 'top-level indicator' of the pressure of agricultural activity on biodiversity, but note that it is not without its difficulties. First, a number of different approaches have been used to estimate HANPP, and as Haberl et al. (2007) note, studies have produced a range of different results. Firbank et al. (2008) argue that there has to be a consistent approach across different scales. Second, the effects on biodiversity are generally assessed by using quite a narrow range of taxa. The effects of different levels of appropriation may vary considerably between different species groups. They suggest that probably no single indicator of agricultural intensity exists, and that instead it is more appropriate to make a three-fold distinction between: the pressures arising from transformation between non-agricultural and agricultural habitats; changes in 'combinations and arrangements' of crops, livestock and the semi-natural elements found in agricultural landscapes; and changes in management techniques. Using vegetation and breeding bird data derived from Countryside Survey 2000, they show that indicators can be developed to characterise change in relation to the three dimensions they suggest. These indicators partially explain the patterns of change observed for agricultural landscapes in Great Britain, and the different responses by plants and birds. However, it is uncertain how such an approach can be scaled up, and how the link between these indicators and broader measures such as HANPP can be made.

The justification for using HANPP as the basis for assessing the impact of human activity on biodiversity is the so-called 'species-energy hypothesis' (Hawkins et al., 2003) which asserts that there is a positive relationship between available energy and species diversity. It is unlikely that such a relationship is a simple one, as Evans et al. (2005) have demonstrated in relation to the British breeding avifauna.

An examination of the relationships between HANPP and biodiversity is essential if we are to use such measures to make robust assessments of the impact of human management on ecological functioning and to model future trends. One starting point would be to connect this work up with the evolving and parallel debate about the relationship between species diversity and the genera-

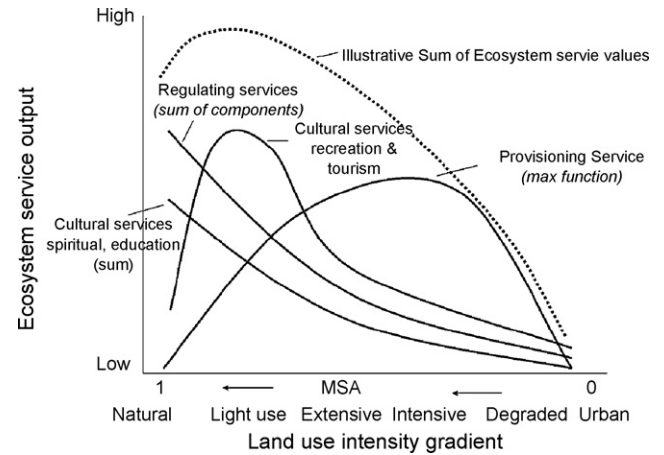


Fig. 4. Hypothetical relationships between land use intensity, Mean Species Abundance Index and the output of ecosystem services (after Braat and ten Brink, 2008). Key: MSA = Mean Species Abundance Index (0 = complete modification of original species pool; 1 = unmodified species pool).

tion of ecosystem services, which focuses on the question of how sensitive the output of ecosystem services is to biodiversity loss (Haines-Young and Potschin, 2010). The underlying assumption is that service output is dependent on the magnitude and intensity of ecological functioning.

Earlier studies, such as those by Schwartz et al. (2000), found little support for a linear relationship between species richness and some measure of ecosystem functioning like productivity, biomass, nutrient cycling, carbon flux or nitrogen use. Instead they argued that the evidence available to them suggested that these functions did not increase proportionally above a threshold that represented a fairly low proportion of the local species pool. However, a more recent meta-analysis by Balvanera et al. (2006) looked at experimental studies involving the manipulation of different components of biodiversity and the assessment of the consequences for ecosystem processes, and suggests the contrary (see also Hooper et al., 2005). They argue that the strength of the relationship between biodiversity and measures of ecosystem function tended to be stronger at the community than the whole ecosystem level.

Questions about the relationships between biodiversity, ecosystem functioning and the output of ecosystem services will continue to challenge the research community as they search for general patterns and responses that can be used to inform policy development. Models such as those proposed by Braat and ten Brink (2008) (Fig. 4) attempt to describe the relationship between gradients of land use intensity, the degree of modification to the native biodiversity and the output of ecosystem services. They need to be tested and refined to take account of different ecological and land management contexts at different geographical scales. These models suggest that the output of service generally peaks at some intermediate level of land use intensity.

The relationships are undoubtedly more complex than Fig. 4 would suggest. Simple gradients of land use intensity are unlikely to explain all variations in biodiversity and the output of ecosystem services unless the effects of fragmentation and the structure of land cover mosaics are taken into account. Although the concept of a functional ecological network is becoming widely accepted as a planning tool (Opdam et al., 2003, 2006), the idea has yet to be extended to look at how variations in structure impact on service output.

Naidoo et al. (2008) have recently made a study of the degree to which current data resources permit the mapping of ecosystem services at global scales. Although they acknowledge that the

evidence base is incomplete, it appears that regions selected for maximum biodiversity seem to supply no more ecosystem services than regions chosen randomly. Moreover, the geographical association between different services, and the connections between ecosystem services and established conservation priorities, appear to show marked variation. They conclude that 'an ambitious interdisciplinary research effort is needed' (Naidoo et al., 2008, p. 9495) if we are to better understand the synergies between biodiversity and service output and the trade-offs that may arise in our efforts to conserve biodiversity and ecosystem services. An illustration of the kind of work that is necessary is provided by the study by Chan et al. (2006). They have demonstrated that at regional scales in the US, a spatially explicit conservation planning framework can be developed that takes account of some of the relationships between land management, biodiversity and the output of ecosystem services. For the Central Coast Ecoregion of California they concluded that because of potential trade-offs between biodiversity and a suite of different ecosystem services, future planning approaches may need to consider 'new geographies,' different from those conventionally used for conservation purposes. They also suggest that these new situations will require us to broaden our current understanding of conservation goals.

Evolving policy perspectives

What are the critical issues for policy makers that emerge from this review of the links between biodiversity and land use? Two key areas can be identified:

- the robustness of the evidence base and the uncertainties involved in framing policy action in the face of rapid change; and
- evolving perspectives on how biodiversity and ecosystem services are valued and what this means for policies that shape land use.

Governments have set the ambitious target of reducing biodiversity loss by the year 2010, through the Convention on Biodiversity. The scientific community has been focused on how to assess progress towards this target and increasingly on what might lie beyond, given that the target is unlikely to be achieved. At the scientific level advances have been made in the design of indicators, such as the measures developed through the UNEP 2010 Biological Indicators Partnership (BIP2010) and European Streamlining Biodiversity Indicators 2010 (SEBI2010) projects (Mace and Baillie, 2007), but much remains to be done (e.g. Commission of the European Communities, 2008). The Biological Intactness Index proposed by Scholes and Biggs (2005), which seeks to provide a global picture of biodiversity change, suffers similar problems to others such as the Mean Species Abundance Index used by the OECD and others (Braat and ten Brink, 2008). These indices are based more on expert opinion than field data, and may significantly under-represent biodiversity loss (cf. Rouget et al., 2006). There is an urgent need to expand the empirical base of such indices, by gathering monitoring data through well-structured sampling designs (Pereira and Cooper, 2006) so that spatially explicit disaggregations can be made reliably, and the trajectories of different regions can be better understood.

If a robust evidence base for policy is to be constructed, biodiversity indicators also have to be better integrated with empirical information on the various drivers of change, and in particular the factors shaping land use, so that better modelling and scenario tools can be developed. The scope of these indicators needs to be expanded so that the consequences of biodiversity change

for ecosystem services and human well-being can be better understood.

We are becoming more aware that the conservation of biodiversity cannot be looked at as an end in itself. Although biodiversity clearly has intrinsic value, it is also apparent that conservation can be justified by more utilitarian arguments that emphasise its role in securing the output of ecosystem services. It is possible that quite different decisions about land use and land cover will be made if these new perspectives on the values associated with biodiversity are factored into decision-making.

The interim report on *The Economics of Ecosystems and Biodiversity* (TEEB) (European Communities, 2008) notes that if we are sustain benefits that ecosystems provide then we may have to rethink the way market systems operate, and try to ensure that the contribution nature makes to human well-being is fully recognised. While market-based approaches involving payments for ecosystem services are likely to shape the management of land and the transactions that surround it, new types of regulatory or legal measures are also likely to be needed to secure the public benefits which arise from land and its associated biodiversity resources. Could measures such as cross-compliance¹ in agriculture be used to ensure the delivery of ecosystem services for which conventional markets do not exist? Can green infrastructure be created and restored via community land trusts supported by some kind of levy on development? It seems that the monetary valuation of biodiversity and ecosystem services will only take us so far in shaping future policy interventions. Monetary values will increasingly have to be considered alongside wider questions of equity, security and resilience, to assess the trade-offs between the different types of output that can be derived from the land under different policy or management scenarios.

Conclusions

This review suggests that the human transformations of land cover and land use are a key driver of the loss of biodiversity and ecosystem services. Coupled with the effects of climate change, these pressures pose significant management and policy questions as we look for strategies to secure a more sustainable future (cf. Schroter et al., 2005).

The key scientific challenges concern the need to develop more comprehensive monitoring systems, and more sophisticated modelling and scenario tools. Significant gaps exist in our ability to understand the relationships between landscape structure, biodiversity and the output of ecosystem services at different spatial and temporal scales. We need to know much more about how qualitative changes in land cover and land use impact upon biodiversity ecosystems services, as well as about quantitative changes in land cover and land use. In the policy arena, the principal questions concern how we can best use such evidence to deal with difficult cross-sectoral issues through a more perceptive, ecosystem approach to decision making (Defra, 2007). We need better ways of valuing the multi-functional character of land and ecosystems and incorporating these characteristics into policy and planning processes. If we are to cope with the pressures of population growth and human induced environmental change, a sound understanding of the relationships between land use and biodiversity is clearly essential. It is also likely to be critical for our future prosperity.

¹ The requirement in Europe that farmers must keep their land 'in good agricultural and environmental condition' to qualify for agri-environmental payments.

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