PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B

Phil. Trans. R. Soc. B (2008) **363**, 777–787
doi:10.1098/rstb.2007.2183
Published online 4 September 2007

Assessing the impacts of agricultural intensification on biodiversity: a British perspective

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Agricultural intensification is best considered as the level of human appropriation of terrestrial net primary production. The global value is set to increase from 30%, increasing pressures on biodiversity. The pressures can be classified in terms of spatial scale, i.e. land cover, landscape management and crop management. Different lowland agricultural landscapes in Great Britain show differences among these pressures when habitat diversity and nutrient surplus are used as indicators. Eutrophication of plants was correlated to N surplus, and species richness of plants correlated with broad habitat diversity. Bird species diversity only correlated with habitat diversity when the diversity of different agricultural habitats was taken into account. The pressures of agricultural change may be reduced by minimizing loss of large habitats, minimizing permanent loss of agricultural land, maintaining habitat diversity in agricultural landscapes in order to provide ecosystem services, and minimizing pollution from nutrients and pesticides from the crops themselves. While these pressures could potentially be quantified using an internationally consistent set of indicators, their impacts would need to be assessed using a much larger number of locally applicable biodiversity indicators.

Keywords: indicator; farmed landscape; habitat diversity; eutrophication; farming systems; biodiversity conservation

1. INTRODUCTION

Agriculture can be conceived as the management of terrestrial ecosystems to divert their productive capacity to serve human needs, and given that these needs will continue to grow during the coming decades, the rate of this appropriation will increase (Millennium Ecosystem Assessment 2005*a*). Inevitably, this diversion reduces the productive potential for non-crop ecosystems and species, both in terms of replacing existing systems and the management of agro-ecosystems. There is a fundamental conflict between the increasing needs of agriculture and the maintenance of non-crop biodiversity at present levels.

Much has been written about the impacts of agricultural practice on biodiversity. There is a great deal of evidence about how farming practices influence species richness and abundance of taxa (Vickery *et al.* 2001; Firbank *et al.* 2003*a*; Fuller *et al.* 2005), about the threats posed by agricultural change to biodiversity (Tucker & Evans 1997; Krebs *et al.* 1999; Petit *et al.* 2001; Tilman *et al.* 2001), and how farming practices can be modified to mitigate these threats and generate benefits (McNeely & Scherr 2003). The biophysical processes relating agriculture and biodiversity are so numerous and interacting that it is difficult to ascribe a particular biodiversity response to an individual agricultural cause. Rather, most biodiversity changes are responses to a suite of agricultural changes that can be regarded together as agricultural intensification (Chamberlain *et al.* 2000) on the one hand, or habitat restoration or abandonment on the other. This complexity means that we lack a clear conceptual model of how agricultural intensification (and by implication, de-intensification, though this is unlikely to be a straightforward reversal) affects biodiversity.

In this paper, we present a strategic view of the different processes that, together, constitute the pressures of agricultural intensification on biodiversity. We then consider indicators for both these pressures and for impacts on biodiversity, here drawn from national data on plant and bird assemblages, and finally, we consider the relationships between the pressures and the impacts to consider appropriate strategies for conserving biodiversity within agricultural landscapes.

(a) *The pressures of agricultural intensification on biodiversity*

It is estimated that, globally, 30% of terrestrial primary productivity is appropriated by humans (Imhoff *et al.* 2004). This level of appropriation varies greatly; in western Europe, it is estimated that levels of human

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One contribution of 15 to a Theme Issue ‘Sustainable agriculture II’.

appropriation of terrestrial net primary production (HANPP) reach 72% (Imhoff *et al.* 2004). The global levels of appropriation will only increase as a result of population and economic growth. This means that the remaining energy will reduce, impacting still further on the numbers, biomass and diversity of non-cropped animals, plants and micro-organisms (i.e. biodiversity, as interpreted in this paper).

The level of HANPP is, potentially, the ideal top-level indicator of the pressure of agricultural intensification on biodiversity. However, there are two problems with its widespread use at the moment. The first is that consistent methods for its estimation are not yet applied at different scales. The second is that while agricultural impacts on biodiversity are usually expressed in terms of different species or species groups, like farmland birds (Donald *et al.* 2001), the impacts of appropriation vary between such taxa. A single indicator of intensification is not sufficient. Here we propose that agricultural intensification can be characterized by three major processes, acting at different scales (albeit with interactions), that can be indicated separately. They are: transformation between non-agricultural and agricultural habitats (including managed forests, an important issue but outside the scope of this paper); the transformation of agricultural landscapes into new combinations and arrangements of crops (including livestock) and semi-natural elements; and the management of these crops to increase their productivity, through breeding, fertilizer use, the introduction of alien species and the control of competitors, predators and parasites (see also Matson *et al.* 1997; McNeely & Scherr 2003).

(i) *Transformation between agricultural and non-agricultural habitats*

Globally, one of the major pressures on biodiversity remains the transformations of natural habitats to agriculture, especially through forest clearance (Jenkins 2003), both alone and interactively with climate change (Thomas *et al.* 2004). However, the transformation is far from one way (Lepers *et al.* 2005). Agricultural land is increasingly being lost owing to urbanization and land abandonment, for economic reasons and owing to loss of productivity (Millennium Ecosystem Assessment 2005*b*). Some transformations between agricultural land and habitats for biodiversity are conceptual, rather than reflecting changes in land management; thus historic farming practices are conserved for their aesthetic, cultural and ecological interest and, by contrast, it is possible that wild grazing animals might be increasingly harvested for human consumption, thus transforming natural ecosystems into agricultural ones.

The transformation of natural to agricultural land was largely completed several centuries ago throughout much of Europe. Indeed, this process is now starting to reverse, partly owing to land abandonment (Petit *et al.* 2001) and partly owing to new programmes of large-scale habitat restoration (Leopold *et al.* 2001).

(ii) *Transformation of agricultural landscapes*

Farmed landscapes have always evolved according to the requirements of the land managers in the context of the social, economic and technological environment (Rackham 1986; Holl *et al.* 2002; Shrubb 2003). However, the rate of landscape change has accelerated globally during the past century. Perhaps the main reason for this is the increasing spatial scale of the human food chain. Many traditional and subsistence farming systems were intended to serve a very local market; such systems were diverse and fine grained, thereby providing variety and resilience at a local level. Fine-grained landscapes are also associated with modern solutions to local-scale sustainable agriculture (including permaculture and agroforestry). However, as many markets became larger and more dispersed, there has been a greater emphasis on farmers specializing on fewer products with consequent reductions in landscape diversity at farm and regional levels. As a result, the diversity of habitats has declined, along with the landscape grain, both resulting in reductions in species diversity (Benton *et al.* 2003).

(iii) *Changes to crop management*

One of the most remarkable human achievements of the twentieth century was that food production more than kept pace with global population increase (Millennium Ecosystem Assessment 2005*b*). This was largely due to increases of crop yield, brought about not least by the use of fertilizers and other inputs, along with the development of new crop varieties. The potential costs to the environment and human health were recognized quite quickly (Culver *et al.* 1956; Carson 1962; Anon. 1969; Newton & Wyllie 1992; Philip 2001), and increasingly these concerns were integrated into agricultural management through regulation and the development of markets for organic and integrated farming produce. Changes in crop management are also technology driven and these have occurred rapidly; for example, the uptake of genetically modified crops in those countries where they are allowed (Fernandez-Cornejo & McBride 2002), the uptake of zero tillage in Brazil (Ekboir 2002) and that of integrated pest management in Asian rice crops (Matteson 2000).

Assessing the impacts of changing farming practice is problematic; there are very many intercorrelated variables to consider (Chamberlain *et al.* 2000). For example, while the environmental impacts of genetically modified herbicide tolerant crops are ultimately the result of new plant breeding technologies, their direct causes are changes to herbicide regimes and farming systems (Firbank *et al.* 2003*c*, 2005). Also, farming practice changes over time, meaning that studies that demonstrate differential biodiversity impacts of farming and land management systems (e.g. Kleijn & Sutherland 2003; Fuller *et al.* 2005) may quickly become outdated.

2. ASSESSING THE PRESSURES ON BIODIVERSITY

It is clear that the pressures of agriculture on biodiversity vary greatly geographically and across scales. They vary not only with time, but the *rate* of disturbance is an important measure of agricultural intensity in some systems. Even so, we suggest that the major aspects of agricultural intensification can be

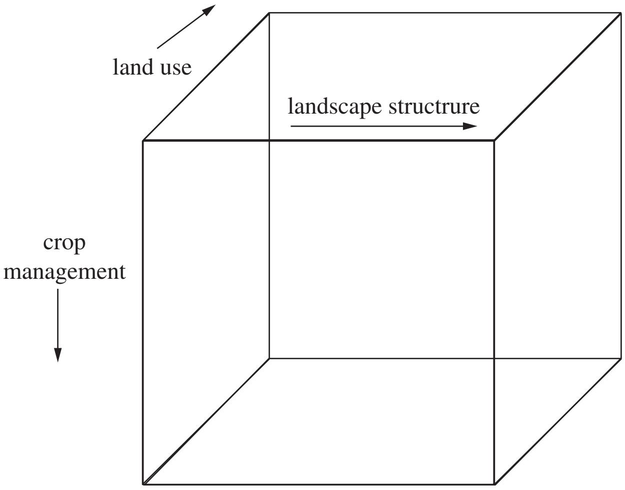

Figure 1. A conceptual model of how agricultural systems can be described according to three dimensions of agricultural intensification, of large-scale land use, of landscape structure and diversity, and the management of crops and livestock at the field scale. It is possible to place any agricultural system within these dimensions for both the pressures on biodiversity and, separately, for the biodiversity states themselves, by using indicators for each dimension. These might include, for pressures, loss of natural habitat to agriculture, losses of landscape diversity and increases in fertilizers and pesticides. For biodiversity states, potential indicators might include populations of species associated with natural habitats, species diversity in agricultural habitats and the trait composition of within-field plant populations. See text for details.

summarized in three dimensions that distinguish between processes at three spatial scales, i.e. large-scale changes in land cover and changes to landscape structure and to land management. Given appropriate indicators, any farming system, of any scale, can be located within these dimensions (figure 1), allowing the comparison of pressures on biodiversity arising from agriculture to be compared directly across space, time and scales. The impacts of these pressures on biodiversity also need to be measured using indicators that are particularly sensitive to the pressures operating at each scale. In principle, it is therefore possible to characterize farming systems in terms of biodiversity as well as the pressures, and to establish a comparative basis for the strategic assessment of the impacts of agriculture on biodiversity.

In this paper, we explore the potential for this approach using data from lowland agricultural areas in Great Britain (GB). These landscapes have showed little large-scale transformation between agricultural and natural habitats in recent centuries, and so we focus on changes in landscape and land management, reducing the number of dimensions to two. Ideally, there would be a unique best indicator for pressure and biodiversity for each dimension, but in practice there are many possible indicators, as discussed below.

We evaluate selected indicators of both pressures and biodiversity using data collected within sample 1 km squares as part of the GB Countryside Survey 2000 (CS; Howard *et al.* 2003; Smart *et al.* 2003*a*). These squares have been classified according to climate, topography and geology into a series of land classes, grouped into environmental zones (Firbank *et al.* 2003*b*). These areas differ in terms of land use, landscape structure and vegetation diversity (Haines-Young *et al.* 2003). We use only data from the three zones that separate the drier, largely arable areas of eastern England from the wetter lowlands of England and Wales, and the more northerly lowlands of Scotland; the three upland zones are not relevant to this study.

(a) *Transformation of agricultural landscapes*

It has been argued that the current decline in farmland biodiversity mainly results from a loss of habitat diversity at multiple spatial and temporal scales (Benton *et al.* 2003). As agriculture has become specialized, there have been both regional- and farm-scale trends for a reduction in the diversity of crops (Hjorth Caspersen & Fritzboger 2002; Shrubb 2003). In Britain, there has been a simplification of agricultural landscapes in the lowlands, with a drastic reduction in the length of linear features over the past 20 years (Petit *et al.* 2003*b*) and a significant increase in the size of parcels used for intensive agriculture (Petit *et al.* 2002).

These changes in farmed landscapes can affect biodiversity in two ways. Firstly, the number and areas of different habitats are affected, and secondly the grain of these habitats is changed. These effects are hard to separate because they are intercorrelated; indeed, a recent Finnish study showed that bird numbers are best described by their joint effect (Heikkinen *et al.* 2004). Moreover, the structure of the cropped landscape (the diversity of crop types and the sizes of fields) can be associated with the diversity of the non-cropped landscape (woods, ponds and hedgerows; Haines-Young *et al.* 2003; Fuller *et al.* 2005), making it difficult to separate effects of agricultural practices from those resulting from the configuration of the landscape as a whole.

(i) *Indicators of landscape structure*

Here we adopt habitat diversity as an indicator of pressure on biodiversity resulting from landscape structure. The data come from the CS2000 field survey of 1 km squares, selected from the lowland environmental zones 1, 2 and 4 of GB on a stratified, random basis and undertaken in 1998 (figure 2; Bunce *et al.* 1996; Firbank *et al.* 2003*b*). Each parcel of land was mapped (minimum mappable unit of 25 m^2) and allocated to one of the 19 broad habitat classes (Howard *et al.* 2003) that include the agricultural classes, arable and horticultural, improved grassland and neutral grassland. For some analyses, these agricultural classes were subdivided into countryside survey main land cover types (Barr *et al.* 1993), six categories for arable and horticultural, four for improved grassland and two for neutral grassland.

For each square, we calculated the Shannon diversity index of the broad habitats, thus taking into account the number of broad habitats and the degree of dominance among them. We then calculated the index as before, but regarding the agricultural main land cover types as distinct categories. This latter measure takes into account variation among crops to a much greater extent.

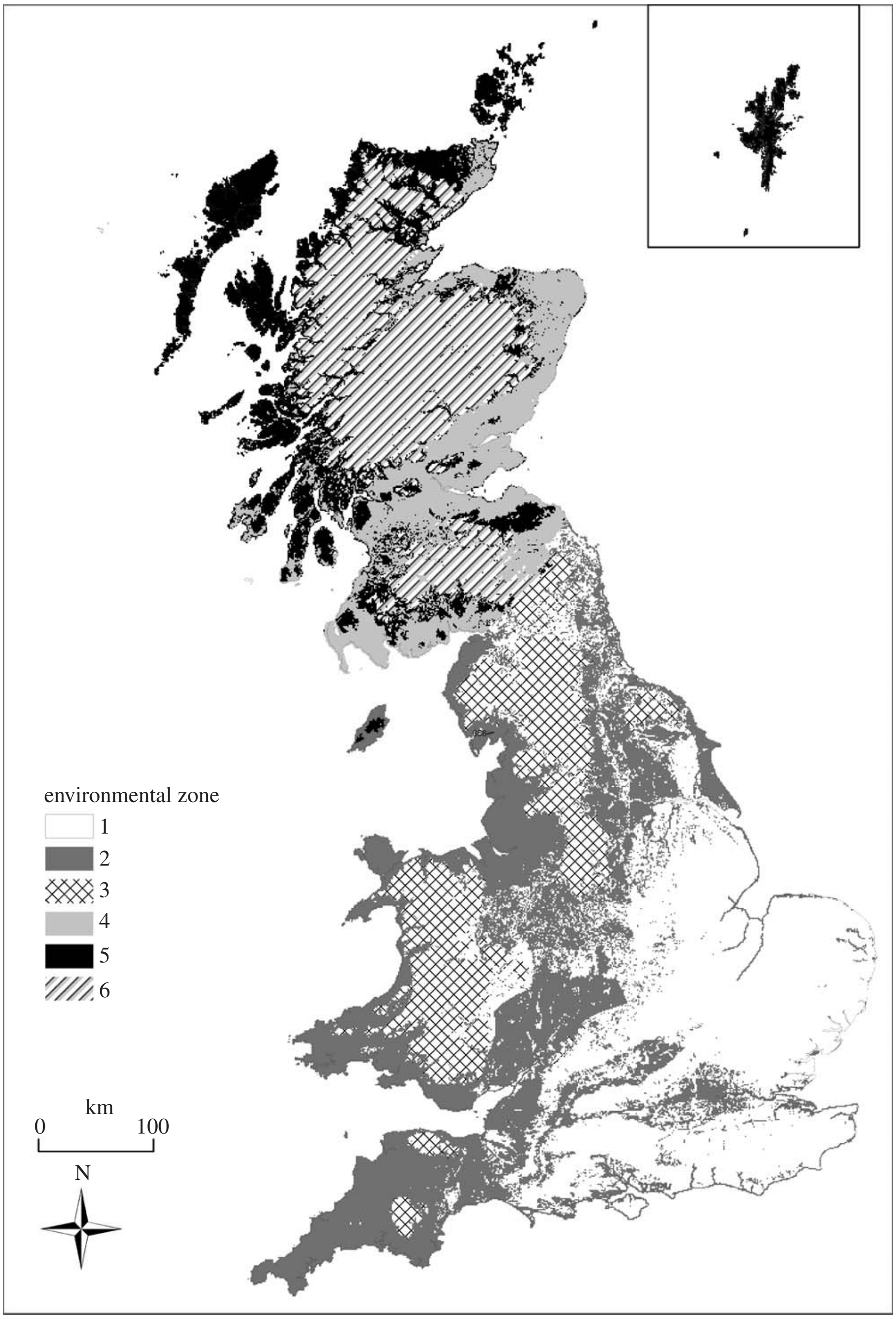

Figure 2. The six environmental zones of GB. These group lands on the basis of climate, topography and geology (Firbank *et al.* 2003*b*). In this paper, we only consider the agricultural lowlands of England, Wales and Scotland, i.e. zones 1, 2 and 4.

(ii) *Indicators of biodiversity*

The most appropriate biodiversity indicators were those that could be generated from data collected from the CS 1 km squares, ensuring correspondence with the landscape data. We selected two that might respond differently to landscape diversity, namely species richness of plants and breeding birds. Plant species richness within CS plots has been used as a national biodiversity indicator in GB (Defra 2004), as it takes into account habitat quality and diversity, and has been recorded in a consistent manner since 1978. Vegetation was sampled within five 200 m^2 plots, placed within the survey square at random within fields or unenclosed land, away from boundaries and linear features (Smart *et al.* 2003*a*). These so-called *X*-plots, therefore, sample vegetation from the broad habitats roughly in proportion to their land cover. Our indicator is the total count of plant species in all five *X*-plots within each square, as recorded in 1998. Bird populations make up the headline biodiversity indicator for UK (Defra 2005); here we use species number of all birds found within the CS squares. Breeding birds were recorded along four parallel (as far as possible) 1 km transects within 176 of the sample 1 km squares (Wilson & Fuller 2002).

(iii) *Results*

The two landscape indicators were regressed against the two biodiversity indicators. All four analyses

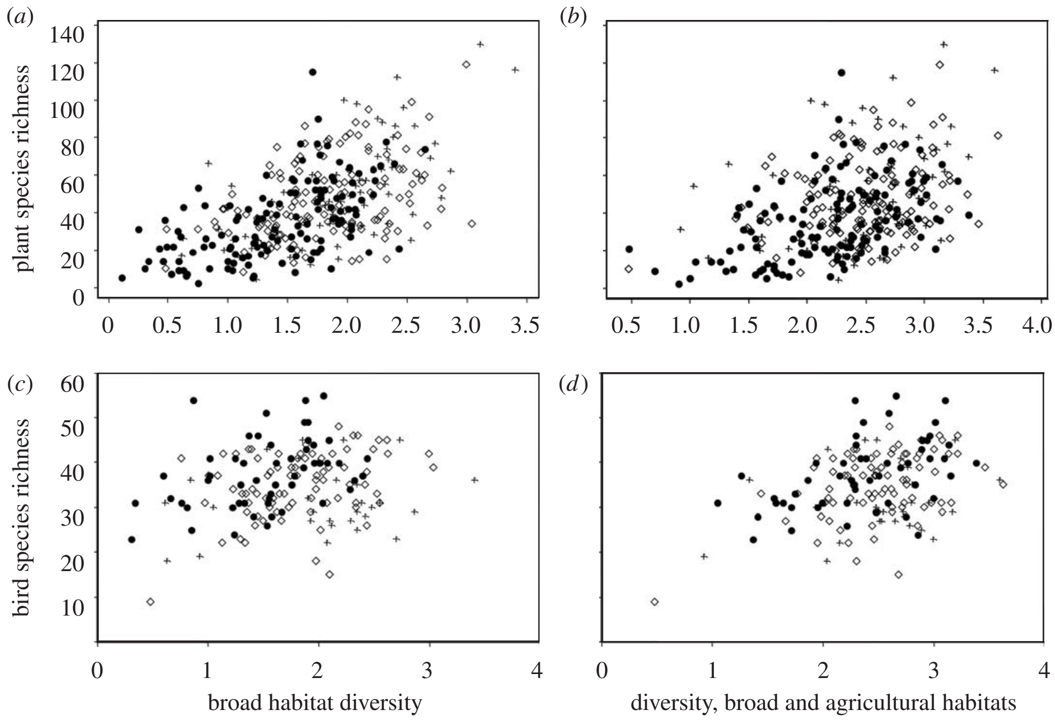

Figure 3. Relationships between indicators of landscape diversity—(*a*,*c*) the Shannon diversity index of broad habitats only and (*b*,*d*) the Shannon diversity of broad habitats, subdividing agricultural broad habitats into main land cover types—and biodiversity ((*a*,*b*) species richness of plants on sample plots located in fields and unenclosed land, (*c*,*d*) species richness of breeding birds as sampled within transects) for the three environmental zones of lowland GB (black circles, zone 1; open diamonds, zone 2; and crosses, zone 4), as recorded within sample 1 km squares.

revealed statistically significant, positive relationships between landscape structure and species richness, though relationships were often weak (figure 3). The species richness of plants was better accounted for by the diversity of broad habitats alone than by taking into account the diversity of main land cover types for agriculture ($r^2=32.2$ and 16.4%, respectively). The sample squares from environmental zone 1 tended to have both lower broad habitat diversity and species richness than squares from the other zones.

By contrast, the species richness of breeding birds was better related to the diversity measure that included both broad and agricultural habitats ($r^2=3.3$% for broad habitats only, 10.5% otherwise). Bird species richness in environmental zone 1 was not lower than in the other zones.

(b) *Changes to crop management*

Although there are many pressures caused by different aspects of crop and livestock management on different taxa and habitats, their effects are very difficult to separate because they tend to be applied as whole farming systems. Thus an historical analysis of changing arable agriculture in GB showed that 76% of variation could be explained by a single axis (Chamberlain *et al.* 2000), while 30% of variation in trends among European farmland bird populations could be explained by cereal yield alone (Donald *et al.* 2001). This high degree of intercorrelation of crop management attributes makes it possible for a small number of indicators to account for the pressures of agricultural change at the field level.

For this study, we wished to use a single indicator that was applicable to all agricultural systems and was known to have major impacts on biodiversity. Of all the pressures caused by crop management on biodiversity, the excess of nitrogen and phosphorus is perhaps the most ubiquitous and difficult to manage (Heathwaite *et al.* 1996; Tilman *et al.* 2001; Dalton & Brand-Hardy 2003), even more so than the effects of pesticides that can be much more localized (Roy *et al.* 2003). Excess nutrients result in pervasive impacts on both terrestrial (Smart *et al.* 2003*b*) and aquatic (Carpenter *et al.* 1998) biota (Dalton & Brand-Hardy 2003; Millennium Ecosystem Assessment 2005*b*). Indeed, one of the most consistent changes in British vegetation over recent years is the eutrophication signal in mid-successional plant communities typical of low or moderate productivity (Smart *et al.* 2003*b*). Eutrophication drives the depauperation of plant assemblages through the increase of a small number of potential dominant species that are better able to capitalize on increased nutrient availability (Davis *et al.* 2000; NEGTAP 2001). In lowland Britain, the plant assemblages most susceptible to eutrophication are currently found in semi-natural habitats intimately associated with adjacent, larger areas of agricultural land, especially alongside hedgerows and streamsides (Smart *et al.* 2005). While eutrophication in these areas implies the effects of agricultural nutrient surplus, they may also arise from atmospheric inputs resulting from the burning and processing of fossil fuels in industry and motor vehicles (Vitousek 1994; Matson *et al.* 1997; Carpenter *et al.* 1998). It is therefore important to distinguish the contributions to vegetation change between agricultural inputs and those from other sources.

(i) *Indicator of nutrient inputs*

We used N surplus to indicate nutrient inputs into the environment from agriculture, estimated by the European livestock policy evaluation network (ELPEN) model that uses data including crop area, livestock numbers and fertilizer rates (Wright *et al.* 1999). However, we also had to account for atmospheric deposition of N from other sources. For this, we used estimates of atmospheric deposition, including both agricultural and industrial sources. These values comprise wet deposition and dry deposition of NH_3, NO_2 and HNO_3 from measured concentration fields and a dry deposition model (Smith *et al.* 2000; NEGTAP 2001). Both sources of N were estimated at the 5 km scale, and we assumed a uniform spatial distribution within each 5 km cell. These variables are not available for every year, so we used N surplus data from 1991 and atmospheric N deposition from 1996.

(ii) *Indicator of eutrophication*

The state variable used to indicate eutrophication was the status of vegetation in areas surrounding cropped land, as given by the mean Ellenberg fertility (N) value per CS vegetation plot. Each GB plant species has been allocated an Ellenberg N value ranging from 1 to 9, where each value estimates the position along the abiotic gradient at which each higher plant species should reach maximum abundance (Ellenberg 1988; Hill *et al.* 2000). Thus, as vegetation becomes more typical of more fertile conditions, its mean Ellenberg value increases as a result of changes in species composition and/or relative abundance. These changes are comparable across vegetation types and farming systems.

We selected CS plots sampled in 1998 and located beyond the edge of agricultural habitats (in small semi-natural habitat fragments, road verges and streamsides) so as to exclude direct effects of fertilizer application on fields. Also, we excluded plots dominated by weedy and woody species in order to control for correlations between successional status and Ellenberg N values (Schaffers 2000). Up to 15 plots, 10 sampling linear features (10×1 m) and five sampling small biotopes (2×2 m) were located within the 309 1 km sample CS squares in the three lowland environmental zones (Smart *et al.* 2003*a*).

We partitioned the variation in mean Ellenberg N shared by and uniquely attributable to the two explanatory variables of agricultural N surplus and atmospheric deposition using a mixed-model ANOVA (SAS proc MIXED) following Singer (1998). This analysis also allowed us to separate variance due to differences between plots within the 1 km squares and the more relevant variance in mean Ellenberg values between 1 km squares.

(iii) *Results*

Estimates of both N surplus and N atmospheric deposition were higher in the English lowland zones 1 and 2 than in lowland Scotland (zone 4), and the same was true of mean Ellenberg N scores (figure 4). Both indicators of nutrient levels were significantly associated with mean Ellenberg N score ($p<0.001$). The two explanatory variables together explained 21.7% of the between 1 km square variance in mean Ellenberg N, with 12.4% explained by N surplus alone, 1.3% from atmospheric deposition and a further 8% from their interaction that probably arose from the spatial coincidence between areas of high levels of atmospheric deposition and intensive agriculture. The results provide clear correlative evidence of eutrophication impacts across the British countryside resulting from agricultural nutrients with a smaller contribution from transport and industrial sources, and support the use of N surplus as an indicator of agricultural pressure on biodiversity.

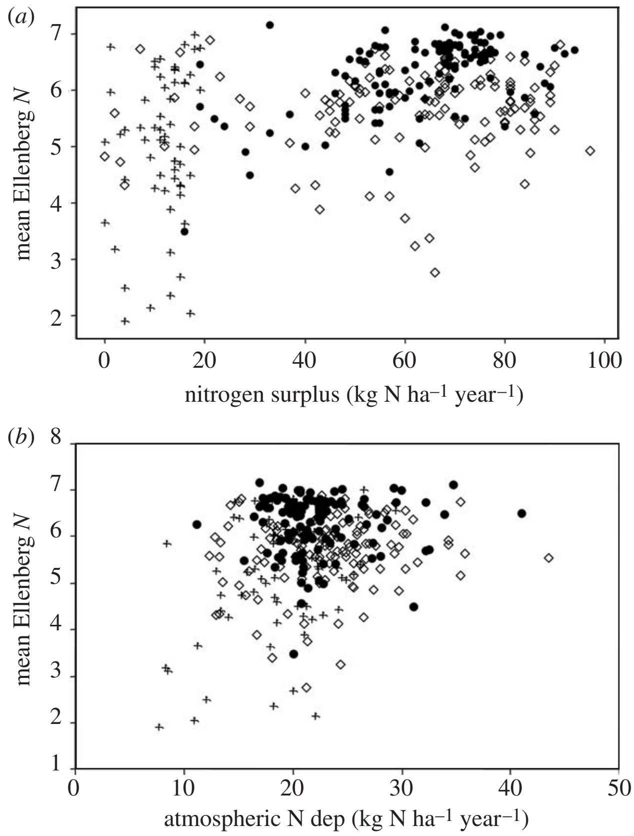

Figure 4. Relationships between indicators of intensity of crop management and nutrient inputs (*a*) N surplus and (*b*) atmospheric N deposition against Ellenberg *N*, meaned across all sample plots within the 1 km square for the three environmental zones of lowland GB (black circles, zone 1; open diamonds, zone 2; and crosses, zone 4).

(c) *Characterizing farmed landscapes in terms of agricultural pressures and biodiversity states*

Having shown that it is possible to relate indicators of agricultural pressure to those of biodiversity in agricultural landscapes, the next question is whether these relationships can be used to characterize those landscapes. Here, the relationship between landscape structure and crop management is given by that between broad habitat diversity and N surplus of individual 1 km squares. This mapping shows strong differences between the environmental zones, with zone 4 (lowland Scotland) having lower nutrient surpluses, while environmental zone 1 tends to have the least diverse landscapes (figure 5).

The relationships between biodiversity indicators of landscape and of crop management varied between taxa; that between plant species richness and mean

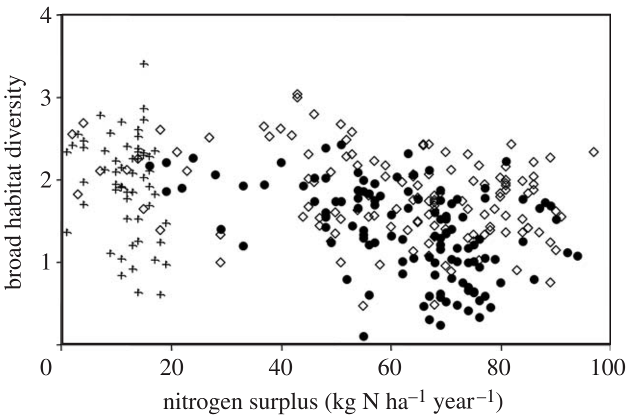

Figure 5. Characterizing 1 km squares of lowland GB in terms of landscape structure (as indicated by the Shannon diversity of broad habitats) and crop management intensity (as indicated by N surplus; black circles, zone 1; open diamonds, zone 2; and crosses, zone 4).

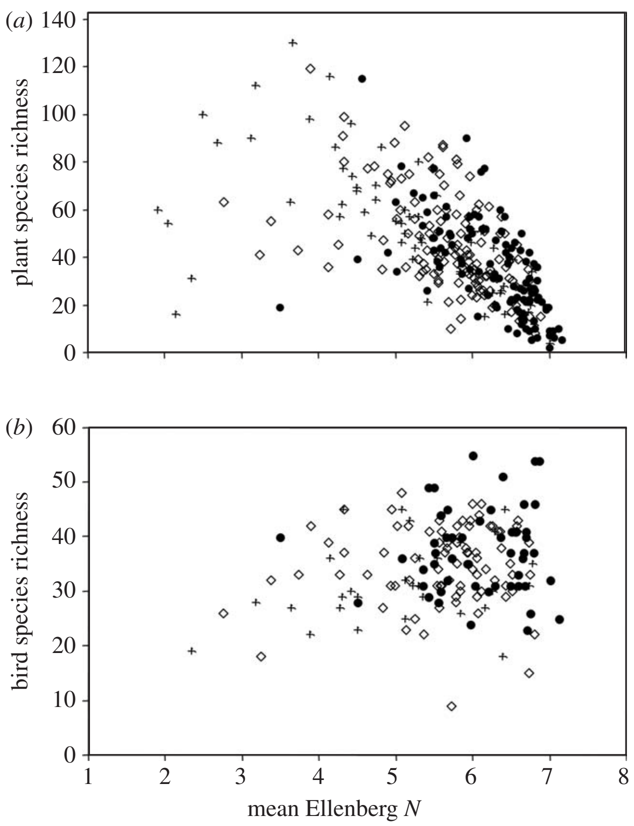

Figure 6. Characterizing 1 km squares of lowland GB in terms of biodiversity indicators of landscape structure. (*a*) Plant species richness per 1 km square and (*b*) species richness of breeding birds per 1 km square and crop management (mean Ellenberg *N* score; black circles, zone 1; open diamonds, zone 2; and crosses, zone 4).

Ellenberg *F* score (figure 6*a*) showed a clear negative relationship, with the least diverse vegetation showing the greatest evidence of eutrophication. By contrast, there was no signification relationship between bird species richness and mean Ellenberg score (figure 6*b*).

3. DISCUSSION

Given the dominant role of agriculture as a driver of change in biodiversity, it is important to establish generic, high-level indicators of both the pressures of agricultural intensification and of the impacts on biodiversity, so that they can be compared across space and time. The ideal indicator is representative of the system, readily measurable, easily understood and analysed (Gregory *et al.* 2005). It is also interpretable within a logical framework. One such model is the driver–pressure–state–impact–response (DPSIR) model, in which the driver refers to the broad social, economical or environment signal; the pressure is the action that impinges on a given state variable, inducing a change or impact, which then feeds back into the drivers by a policy or other response (OECD 1993). Given the range of human activities and environmental processes encompassed within agriculture and the variety of interests to be addressed, it is hardly surprising that a wide range of indicators have been proposed. They have come from national governments (MAFF 2000), international bodies (Commission of the European Communities 2000) and private industry (e.g. www.unilever.com), and contribute to the evaluation of both the sustainability of agriculture and to environmental quality. However, the sheer number of these indicators can make their interpretation more difficult, because they are often so intercorrelated. In general, the fewer the indicators, and the more they can be consistently applied across farming systems, the better. We have proposed that a very small number of indicators are selected to monitor agricultural pressures and their biodiversity responses at different spatial scales; was this approach successful, and could it be applied beyond GB?

(a) *Can impacts of agriculture on biodiversity be assessed in only three dimensions?*

At a global scale, the pressure of habitat loss to agriculture remains a major threat to biodiversity (Tilman *et al.* 2001) and is readily monitored using remote sensing, and the impacts can be assessed at levels from ecosystems to genetic diversity (Heywood & Watson 1995). In GB, this process is, if anything, going into reverse but not yet at levels detectable within CS data, leaving our study with only two dimensions representing the landscape and field scales. It proved possible to characterize individual 1 km squares, and the broader regions in which they are found, by quantifying indicators for processes at these two scales. In general, squares located in environmental zone 1, dominated by arable agriculture, showed the greatest pressure of agriculture on biodiversity, by combining a less diverse landscape with more intensive crop management (figure 5). These indicators could be applied globally, if there were both validated N surplus models and a consistent international habitat classification that includes both semi-natural and cropped habitats.

The vegetation indicators were highly correlated with these pressures, as expected. Habitat diversity accounted for 32% of variation in plant species richness and N surplus accounted for at least 12% of variation in mean Ellenberg *N* score. By contrast, species richness of breeding birds was far less sensitive to these pressures. The reasons are simple, that while plant assemblages respond closely to nutrient inputs and habitat diversity at the scales studied within CS, bird assemblages are more sensitive to changes within the farmed landscape not well captured by these indicators, especially the availability of food resources within and adjacent to the fields: birds respond to agricultural pressures at both the landscape and crop management scale in a more integrated way.

Such differences suggest that it is unrealistic to indicate biodiversity responses to agricultural change with a single taxon at each scale. Other studies have frequently shown that the responses of different taxa to agricultural landscape structure and management are not well correlated (Dauber *et al.* 2003; Petit *et al.* 2003*a*; Burel *et al.* 2004). Agri-biodiversity indicators are also sensitive to factors at different spatial and temporal scales, not least large-scale gradients of species numbers (Kivinen *et al.* 2006). Responses to land transformation and landscape change are not clearly separated, rather they form a gradient; species with small area requirements can persist in highly fragmented patches of habitat in agricultural landscapes which would be too small to maintain populations of species with large home range (Vos *et al.* 2001). Perhaps the only appropriate indicators of biodiversity responses to agriculture for comparisons across large geographical areas are those that respond to changing patterns of ecological traits, such as Ellenberg values or mobility patterns, rather than on species occurrence or richness. Such indicators may prove expensive to develop, given that traits need to be established for individual species and can vary within different parts of their range (Hill *et al.* 2000).

(b) *Implications for maintaining biodiversity within agricultural landscapes*

The value of a strategic approach to assessing the impacts of agricultural intensification is that it should inform the development of policy responses, especially strategies for conserving biodiversity in the context of agricultural change. Just as the pressures of agricultural change can be described at different scales, so can the strategies for biodiversity conservation.

Any global strategy must recognize the importance of maintaining large areas of natural habitat, especially in global biodiversity hotspots. It has been suggested that this is served better by reducing the land requirement for agriculture by increasing productivity per unit area than by adopting less intensive agriculture (Green *et al.* 2005). It can also be argued, however, that a more effective of strategy would be to minimize losses of existing agricultural land to land degradation (salination, desertification, etc.) and urbanization. Conversion of land to forest, natural areas and even abandonment are less of a problem, because the loss from agriculture could be reversed, while the replacement land covers can provide other important ecosystem services such as water regulation and carbon storage. Agricultural land is too valuable a global resource to fritter away, especially in the context of potential sea level rise on the one hand and increasing global demand for food, fibre, biofuels and novel crops on the other.

At the landscape level, biodiversity can be supported by maintaining the diversity of habitats, both cropped and uncropped, at a diversity of spatial and temporal scales (Benton *et al.* 2003). Such diversity can be achieved either as an intrinsic part of the agro-ecosystem (e.g. rotations, agroforestry and fallows), as additional land uses interspersed among the fields (e.g. small woodlands for game shooting (Duckworth *et al.* 2003)), or as part of on-farm conservation measures. The range of species that can be supported depends upon the diversity and areas of habitats and their quality, spatial arrangement and history. For many species, the most important factors are habitat quality and area. Spatial arrangement of habitats is important for taxa of limited dispersal and with habitat requirements located in a very patchy way in the landscape. Such species include ancient woodland plants (Petit *et al.* 2004), butterflies of grassland patches (Hanski 1999) and some woodland birds (Opdam *et al.* 2003). For some other species, dispersal is so slow that distribution patterns are best explained by historic, rather than by present, distributions of habitats (e.g. the carabid beetle *Abax parallelepipedux*, Petit & Burel 1998). It has been suggested that habitat diversity can be enhanced through agri-environment schemes (Benton *et al.* 2003) and wider use of organic farming (Fuller *et al.* 2005). Both approaches assume a reduced level of agricultural productivity per unit area, consistent with at least some scenarios of European land use change given continued policy support (Rounsevell *et al.* 2006). A more sustainable approach to habitat management might be to redesign rural landscapes to achieve multiple production of ecosystem services; of food and water management, of biofuels and fibre crops, and as sources of pollinators and natural enemies of pests (McNeely & Scherr 2003; Millennium Ecosystem Assessment 2005*b*). Such multifunctional land management, that integrates forestry, energy generation, flood control and food production, would be sensitive to the local environment, and so generate diverse landscapes that would attract their own suites of species (Firbank 2005).

At the field level, the use of N surplus as a pressure highlights the importance of pollution arising from inefficient crop management. Biodiversity benefits can be obtained without sacrificing production by controlling losses of nutrients and pesticides through, for example, developments in precision agriculture (Godwin *et al.* 2003), the use of buffer areas (Correll 2005; Bradbury & Kirby 2006) and increasing the nitrogen use efficiency of livestock by manipulating their feed (Winters *et al.* 2004). It is also possible to benefit biodiversity by allowing more non-crop species to persist within the field, for example, by allowing weeds and invertebrates to build up in some parts of the field (Sotherton 1991; Haysom *et al.* 2004), or at some times of the year (Moorcroft *et al.* 2002).

(c) *Conclusion*

It is inevitable that agricultural change will continue to affect land cover, landscape structure and crop management in ways that impinge on different taxa, though rates and timing of change are hard to predict. These changes are complex, and need detailed monitoring and ongoing research. However, there is case for a high-level assessment of pressures that arise from agriculture using a small number of indicators of agricultural pressure that could be recorded globally, allowing comparisons from place to place and from time to time. The effects of these pressures would need to be assessed using a larger number of more locally relevant indicators of biodiversity.

This work was funded by Defra. We thank many colleagues, and two anonymous referees, for discussions of the ideas included in this paper.

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