SUSTAINABLE USE OF BIODIVERSITY:

A REVIEW OF CURRENT RESEARCH IN THE UK

BioSTRAT Project

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Background

This review is being carried out to enable BioSTRAT, a EU-funded project in support of EPBRS, to compile a European review of the topic prior to the next meeting of the EPBRS in Germany in May (and the e-conference that will precede it in March-April).

Although a comprehensive review of the topic would be useful, this document should be regarded as an introduction to the discussion, outlining where there has been research and implicitly or explicitly identifying where gaps exist. It should be noted that the final outcome of this process will be a short set of recommendations from the German EPBRS meeting.

The topic of the German EPBRS meeting is *Shaping European Biodiversity Research for a Sustainable Europe*. The organisers acknowledge that this is a huge topic and intend to focus on "aspects of sustainable use of biodiversity in the wider EU countryside, which means mainly outside Natura2000 areas". They seek to link the reviews to the Action Plan of the EU COM (2006) 216 on "Halting the loss of biodiversity until 2010 and beyond", proposing three priority research areas:

A. High value farmland (and forest) areas
B. CAP implementation on biodiversity, including aspects of cross-compliance
C. Indicators and monitoring schemes

And also suggest the following:

D. Mainstreaming biodiversity aspects into different strategies and plans: Rural development plans, Less favoured area regime, Strategic Environmental Assessment
E. Genetic diversity of crop varieties, livestock breeds and races
F. Risk areas for soil biodiversity
G. Reduction of main pollutions and its impact on biodiversity
H. Inclusion of biodiversity aspects into flood risk management plans
I. Development of cohesion and structural funds towards directly of indirectly benefiting biodiversity
J. Participation of civil society in Environmental Impact Assessment (EIA) and other processes
K. Ecological coherence and functioning and its inclusion into spatial planning

The following is a preliminary assessment, focussing on the "priority" areas proposed. Note that the review is intended to cover work done in the UK but it has been expanded slightly to include collaborative work including UK researchers and research done outside the Europe by UK researchers.

SUSTAINABLE USE OF BIODIVERSITY

1. INTRODUCTION

This review outlines some of the research relevant to the sustainable use of biodiversity either done in the UK or done, usually collaboratively, by UK scientists elsewhere in Europe or outside Europe. The research follows the suggestions of the organisers of the next meeting of the EPBRS to address particular topics. Some of these relate very closely to the sustainable use of biodiversity but others relate much more to the impact of agricultural and other practices on biodiversity.

2. DESCRIPTION OF MAIN FINDINGS OF THE SELECTED STUDIES

A. High value farmland and forestry

Agricultural intensification is known to be a major pressure on biodiversity (Petit & Elbersen, 2006) although few studies have monitored its impact over time. One notable exception is the monitoring of insects at Rothamsted Experimental Station since 1933 (Southwood et al., 2003). More recently, regular surveys of the UK countryside have provided evidence of change (Black *et al.*, 2003; Cooper *et al.*, 2003; Haines-Young *et al.*, 2003).

Bird species have been notably affected by agricultural intensification (Donald et al., 2002; Donald et al., 2006) and their decline is probably due to a decline in the availability of invertebrate and seed food (Wilson et al., 1999). Plant diversity also shows a decline linked to agricultural intensification (Hodgson et al., 2005). As concern for the impact of agricultural practice on biodiversity as grown, so too has the appreciation of low-intensity farming systems (Bignal & McCracken, 1996) and the way in which particular agricultural practices influence biodiversity.

Agricultural practices, such as drainage and weed management, are key to the survival of many species (Cooper et al., 2005; Killeen, 1998; Mauchline et al., 2005). The type of field margin, for example, influences bumblebee diversity (Pywell et al., 2006): bumblebee abundance in a two-month period in summer was significantly higher on pollen and nectar margins (86 +/- 14 bees per 100 m) compared with wildflower margins (43 +/- 14), mature grass margins (6 +/- 14), recently sown grass margins (8 +/- 4)and the cereal crop (0.2 +/- 0.1). The plant species present in the field margins also affects bumblebee presence (Pywell et al., 2005b). Field margins have less of an influence on other taxa such as Coleoptera and spiders, which benefit more from hedgerows (Pywell et al., 2005a), although recent research highlights the potential for managing field margins to promote beetle diversity (Woodcock *et al.*, 2007). Another feature of agricultural landscapes in the UK, green lanes (unmetalled tracks between fields, either below or above field level, with grass banks, hedgerows or stone walls on either side) promote butterfly diversity (Dover et al., 2000). More generally, habitat heterogeneity appear to be the key to promoting biodiversity conservation in agroecosystems (Benton et al., 2003; French & Picozzi, 2002). Agricultural practices, such as manure application, can also promote dispersal of high conservation value species (Edwards & Younger, 2006).

Although most attention has been placed on cropping systems, high nature value grazing systems should not be overlooked (Evans et al., 2003). Research on grazing has included studies on birds (Loe *et al.*, 2007), voles (Evans *et al.*, 2006), carabids (Cole *et al.*, 2006; Dennis *et al.*, 2004) and other invertebrates (Tallowin *et al.*, 2005; Woodcock *et al.*, 2005).

Research in the UK and elsewhere has increasingly shown how agricultural practice affects individual species or groups of species. However, the development of high-nature-conservation-value farming systems requires a broad understanding of the influence of farming systems and their impact on a range of taxa (McCracken & Bignal, 1998).

Until now the concept of High Nature Value (HNV) has been mainly applied to farming areas. The concept of HNV farming in Europe originated in work undertaken in the early 1990s and in reports produced at the time by the Institute for European Environmental Policy (IEEP), the Joint Nature Conservation Committee (JNCC) and the European Forum for Nature Conservation and Pastoralism (EFNCP) (see, for example, Baldock *et al*., 1993; Beaufoy *et al*., 1994; Bignal & McCracken, 1996; 2000). The first published study who addressed this at a European wide scale was *The Nature of Farming* (Beaufoy *et al.* 1994; Bignal & McCracken 1996, 2000). This study showed for 9 countries spread over Europe that particular farming systems were important in maintaining nature value over much of the wider European countryside and it gave the broad characteristics of HNV farming systems including their broad geographic locations. It concluded that most HNV farms were located in the more marginal areas and on the poorer lands, these included mountain areas, marshlands and the more arid areas of Europe. Differences in data availability dictated that different approaches were taken in this *Nature of Farming* study across all nine countries but in general this involved drawing information from a combination of sources, such as maps of specific agricultural land use or the extent of Less Favoured Areas, agricultural statistics, the location of semi-natural habitats and/or known characteristics of specific systems. It meant that it was not able to map all HNV areas in a consistent way.

Further refinement and development of the approach taken in the *Nature of Farming* was necessary in order to clarifying what type of indicator(s) may be best to help establish where HNV systems occur across Europe and how to target these in European rural development policy. In a tender contract with the European Environment Agency a follow up study was then performed in which a first HNV farmland indicator for the whole EU was developed (Andersen et al., 2003). In this study HNV farmland areas were identified using CORINE land cover, Farming data from FADN and bird distribution data. In this study High Nature Value farmland was defined as those areas in Europe where agriculture is a major (usually the dominant) land use and where that agriculture supports, or is associated with, either a high species and habitat diversity, or the presence of species of European conservation concern, or both. From this, 3 types of HNV farmland were distinguished:

- Type 1: Farmland with a high proportion of semi-natural vegetation;

- Type 2: Farmland dominated by low intensity agriculture or a mosaic of semi-natural and cultivated land and small-scale features;

- Type 3: Farmland supporting rare species or a high proportion of European or World population;

Based on the Andersen et al. study, the EEA and UNEP published a Joint Message (EEA 2004), presenting a preliminary map of HNV farmland for the EU (EEA, 2004), and from this the IRENA indicator 26 was also developed (EEA, 2005). In both publications the HNV farmland indicator used was based on the CORINE Land Cover (CLC) data. Even though CLC is the best EU wide source of land cover data, Andersen et al. (2003) highlighted the many limitations of using such an approach to identifying High Nature Value Farmland. For example, the minimal unit with the CLC is 25 ha which means that may small scale feature of HNV are not recognised or identified. In addition, CLC provides no indication as to the condition of any of the farmland habitats recognised nor indeed as to whether those habitats are still under any form of farmland management. Given the potential difficulties of operationalising the HNV approach across Member States, DG Agriculture is currently (February 2007) funding a study of HNV indicators with a view toprovide clearer guidance to member states as to what is meant by HNV, how they can go about recognising HNV areas, features andd farms and how they could consider evaluating whether HNV farms (and forests) are receiving appropriate support under the new Rural Development Plans. This DG Agriculture study is led by the Institute of European Environmental Policy and involves input from a range of European experts.

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Commercial forestry has led to the loss of areas of high conservation value forest,but rhas begun to restore native woodlands such as Atlantic oakwoods and lowland mixed broadleaved woodlands, that were damaged by the creation of plantations using non-native species. Research is, however, underway to restore habitats damaged by under-planting with non-native species such as Atlantic oakwoods (Truscott *et al.*, 2004), and to evaluate different techniques for the removal of non-native species to secure and enhance ecological processes (Thompson and Hope, 2005).

Plantation forests are not necessarily poor in biodiversity. Research comparing plantation with native forests, for example, has shown that plantations support a range of species assemblages of high conservation value (Eggleton *et al.*, 2005); Humphrey et al., 2003; Humphrey et al., 2000; Humphrey et al., 2002). For birds, tree age and species composition is important in determining their abundance and diversity (Donald et al., 1998).

Although the management of a field or an individual forest stand is important for biodiversity, a broader perspective is important. In an agricultural context, the farmland mosaic in important for species such as butterflies (Ouin et al., 2004). Taking a landscape view, the pattern of woodlands in the landscape is important in conserving both forest and non-forest species (Hope et al., 2006).

UK researchers have also been active outside Europe in, for example, identifying the type of forest cover in agricultural landscapes that benefits biodiversity (Harvey et al.,

2006) and the value of different types of forest plantations for biodiversity (Chey et al., 1997; Lawton et al., 1998; Stork et al., 2003; Watt et al., 2002).

Clearly, the promotion of high nature value agriculture and forestry means that environmental (and perhaps social) goals are prioritised over short-term economic goals. Nevertheless, there may be situations where high biodiversity leads to agricultural benefit. In a study of herbage production, for example, yield was greater in hay meadows grown from seed mixes of 25-41 species compared with 6-17 species (Bullock et al., 2001). In another study, the risk of bovine tuberculosis was found to be lower on farms considered to be favourable to wildlife conservation (Mathews et al., 2006). Agricultural yields are, however, more often likely to be reduced by practices that promote biodiversity (Marriott et al., 2004). Nevertheless, more generally, there is evidence that the value of ecosystem services diminishes with biodiversity and ecosystem loss (Balmford et al., 2002; Turner et al., 2003). Furthermore, novel practices, such as genetically modified herbicide tolerant crops, may offer benefits to biodiversity within intensive systems (Pidgeon et al., 2004) as may the development of organic agriculture (Cobb et al., 1999; Wilson et al., 1997).

Although this review focuses on agricultural and forest habitats, other managed habitats can contribute significantly to biodiversity, e.g. urban gardens (Gaston et al., 2005), although these environments may be subject to heavy pollution (Fountain & Hopkin, 2004). And although agricultural and forest issues have been largely considered separately above, silvoarable systems have and will form an important environment in many parts of Europe (Eichhorn et al., 2006).

B. Influence of national plans of CAP implementation on biodiversity, including aspects of cross-compliance

In the UK, targeted habitat creation under agri-environment schemes has been shown to benefit butterflies, particularly the use of pollen and nectar seed mixtures under the Countryside Stewardship Scheme (Pywell et al., 2004). In contrast, a five year study of the Arable Stewardship Pilot Scheme in the UK showed that most bird species studied were not beneficially affected (Stevens & Bradbury, 2006).

Despite concern about the implementation of agri-environment schemes (Berendse et al., 2004; Critchley et al., 2004; Kleijn et al., 2006; Kleijn & Sutherland, 2003), including the set-aside schemes of the 1990s (Critchley & Fowbert, 2000; Firbank et al., 2003), many researchers conclude that the development of such schemes can benefit biodiversity (Bradbury et al., 2000; Vickery et al., 2004). The known benefit of “winter bird crops” or over-winter stubble fields and their management, for example, could be incorporated in such schemes (Buckingham et al., 1999; Henderson et al., 2004; Whittingham et al., 2006) and agri-environment schemes could provide a policy basis for conserving specific habitats (Franco & Sutherland, 2004) or for habitat restoration (Thompson et al., 1999; Wilcox, 1998). It has also been argued that schemes designed to encompass large congruent areas are more likely to be successful that a series of smaller habitat patches (Whittingham, 2007). Agri-environment schemes together with provisions under the Water Framework Directive could also, for example, benefit farmland birds by reducing the impact of diffuse pollution (Bradbury & Kirby, 2006).

Research on the wider economic (Boardman et al., 2003; Evans et al., 2003), social (Hounsome et al., 2006; Pretty & Smith, 2004; Siebert et al., 2006) and policy (Winter, 2000) implications of agri-environment schemes is also needed. The behaviour and attitudes of farmers and other land managers towards the implementation of agricultural policies (Lobley & Potter, 1998; Macdonald & Johnson, 2000; Morris, 2006; Morris & Evans, 2004; Morris & Potter, 1995; Siebert et al., 2006; Skerratt & Dent, 1996; Wilson & Hart, 2000, 2001) are important in understanding the successful uptake of such policies. Examples include national (Fish et al., 2003), cross-European studies (Siebert et al., 2006) and transatlantic comparisons (Robinson, 2006). Participation in current schemes may, for example, be influenced by management history (Riley, 2006). Agri-environment schemes also need to be considered in comparison to alternative approaches to biodiversity conservation (Mattison & Norris, 2005; Wood & Lenne, 2005), such as protected areas and game management (Robertson et al., 2001; Stoate, 2002), and placed in a regional context (Tilzey, 2000). Furthermore, potentially conflicting goals and practices cannot be ignored (Banks & Marsden, 2000; Hodgson et al., 2005; Rickard, 2004; Young et al., 2005). There certainly remains a clear need to avoid the misuse of "environmental objectives… to provide financial support to farms that are of intrinsically low biodiversity and nature conservation value" (Bignal, 1998).

C. Indicators and monitoring schemes

Previous meetings of EPBRS have considered monitoring and indicators in detail. A few studies may however, be mentioned because their intention was to assess the value of agri-environment schemes and other initiatives designed to promote high value agriculture or forestry. These include mollusc monitoring (Killeen, 1998), the biodiversity associated with agri-environment schemes (Pywell *et al.*, 2006) and forest biodiversity (Dudley *et al.*, 2005; Humphrey *et al.*, 1999; Jukes *et al.*, 2002).

In keeping with the 'open' nature of the marine ecosystem, approaches to assessing the wider (in-combination or cumulative) consequences of human activities are being addressed by various UK governmental and research establishments in partnership with industry. This is in response to domestic and international regulatory needs and, notably (since this encapsulates the 'direction of travel'), the developing EU Marine Strategy. Marine monitoring along with the use of supporting indicators has, for many years, accompanied the majority of licensed activities, for example, those of the construction and extractive industries. The present scientific challenge is to enhance the capability to integrate locality- or sectoral-specific assessments of the consequences for marine biodiversity with those conducted on a broader scale, exemplified by the sea-wide monitoring of fish stocks and communities. It may be summarised as the search for harmony between 'top-down' and 'bottom-up' approaches. Examples of its practical implications include the search for biodiversity measures which overcome sampling and biogeographical dependencies, the requirement for continuity in sampling and analytical practices especially for long-term monitoring, and recognition of the importance of reference points in space or time (e.g. Hardman-Mountford *et al.*, 2005; Leonard *et al.*, 2006; Rees *et al.*, 2006; Rogers & Greenaway, 2005).

D. Mainstreaming biodiversity aspects into different strategies and plans: Rural development plans, Less favoured area regime, Strategic Environmental Assessment

Several examples are given above on how biodiversity aspects are being mainstreamed in agricultural and other policies. Declines in grassland biodiversity, for example, are being addressed through rural policies (Marriott et al., 2004).

Not only are native pine forests (which are included in UK Biodiversity Action Plan targets and the delivery of Scottish Forestry Strategy and Scottish Biodiversity Strategy objectives) important for biodiversity but they also contribute to rural development and economic prosperity (McIntosh, 2006). Management of wild herbivores, which are themselves an important influence on biodiversity (Hester *et al.*, 2000; Kirby, 2001; Stewart, 2001), are an important source of income to local communities (Gordon et al., 2004).

Strategic Environmental Assessment has included consideration of the impact of road development on lowland heathland, the Dartford warbler (Sylvia undata) and the sand lizard (Lacerta agilis) (Treweek et al., 1998), and the farming of Atlantic salmon (Thompson et al., 1995).

E. Genetic diversity of crop varieties, livestock breeds and races

Research on wheat diversity in the UK has shown no evidence of a narrowing of genetic diversity since the 1930s (Donini *et al.*, 2000): modern varieties encompass most of the diversity found in earlier decades. The genetic diversity of barley in the UK has also been shown to have been maintained in recent decades, although there is a concern that a reduction in the number of independent breeding programmes would have a negative impact on the maintenance of genetic diversity (Koebner *et al.*, 2003). At a global scale, current approaches to on-farm conservation are thought to be inadequate (Wood & Lenne, 1997).

The conservation of livestock breeds is critical for the provision of resources for the future of agriculture, especially in the developing world (Hall & Bradley, 1995). Recent years have seen advances in the methods used to investigate of genetic diversity in livestock breeds (Freeman *et al.*, 2006), to assess the risk of extinction of particular breeds (Gandini *et al.*, 2004) and to use naturally-occurring genetic variation in breeding programmes (Bishop & Woolliams, 2004). An assessment of the extinction risk of local breeds of pig, for example, identified priorities for cryopreservation (Ollivier *et al.*, 2005).

Related socio-economic research includes the development of methods for economic valuation of farm animal genetic resources (Drucker *et al.*, 2001). The number of livestock breeds in an area has been found to be related to the size of the human population and its genetic diversity (Hall, 1996).

Research outside Europe has included work on genetic diversity of Asian water buffalo (Barker *et al.*, 1997), pigeonpea *Cajanus cajan*, an important subsistence crop in India (Burns *et al.*, 2001) and on timber species such as Caribbean pine (Zheng & Ennos, 1999), cedars and mahoganies (Cavers *et al.*, 2005a; Cavers *et al.*, 2005b; Cavers *et al.*, 2003a, 2003b; Lemes *et al.*, 2003; Lowe *et al.*, 2003; Newton *et al.*, 1996; Newton *et al.*, 1993).

Priorities include the development of better approaches for on-farm conservation of genetic resources (Wood & Lenne, 1997).

F. Risk areas for soil biodiversity

There are many sources of risk to soil biodiversity and its functioning, including GM plants (Cowgill et al., 2002; Lilley et al., 2006; Mulder et al., 2006), nitrogen deposition (Bragazza et al., 2006), phthalates (Cartwright et al., 2000), zinc and other metals (Davis et al., 2004; Ellis et al., 2001), herbicides (Bromilow, 2004), insecticides (Boucard et al., 2004), oil (Bundy et al., 2004) and other contaminants (Semple et al., 2003), soil tillage practices (Holland, 2004), overgrazing (Sansom, 1999) and forest fragmentation (Sousa et al., 2006). All components of soil biodiversity are at risk although those thought to be most at risk are species-poor macrofaunal shredders of organic matter, soil bioturbators, specialized bacteria such as nitrifiers and nitrogen fixers, and fungiforming mycorrhizas (Brussaard et al., 1997). Methods to measure risks to soil biodiversity include bait lamina to assess the effects of metals on the abundance and diversity of decomposers such as earthworms, isopods, molluscs, myriapods, springtails and mites (Filzek et al., 2004), soil biomarker test systems (Weeks et al., 2004), earthworms for assessing ecological function (Smith et al., 2005), soil quality indices to assess the success of the remediation of contaminated soils (Dawson et al., 2007), Fourier transform infrared (FT-IR) spectroscopy (Scullion et al., 2003) and various biological assays (Paton et al., 2005; Tandy et al., 2005). Decision support systems have been developed to guide management of risks to soil biodiversity (Hossack et al., 2004).

G. Reduction of main pollutions and its impact on biodiversity

The impact of pollutants on soil biodiversity was discussed above. The major source of pollution to terrestrial ecosystems in Europe is nitrogen deposition (Bobbink et al., 1998; Britton & Fisher, 2007; Dalton & Brand-Hardy, 2003; Jones et al., 2004; Phoenix et al., 2006). Along with sulphur pollution, there has been significant research on the concept of critical loads (Britton & Fisher, 2007; Bull et al., 2001; Ling, 2003; Phoenix et al., 2006). Other pollutants include atmospheric metal pollution (Branquinho et al., 1999) and ozone (Davison & Barnes, 1998).

In many cases, other factors influence the impact of pollutants on biodiversity. Grazing, for example, can amplify the impact of nitrogen pollution in montane ecosystems (Hartley & Mitchell, 2005; van der Wal et al., 2003). Climate change is also likely to interact with pollution in influencing biodiversity (Britton et al., 2001).

Sulphur pollution has caused widespread acidification of freshwater lake ecosystems in Europe but this is a decreasing threat (Evans et al., 2001; Ferrier et al., 2003; Helliwell et al., 2003). Nitrogen and phosphorous pollution are now the most significant sources of pollution to freshwater (Ferrier et al., 2001; Helliwell et al., 2001; Johnes, 1996; Mainstone & Parr, 2002). The main pathways of these pollutants have moved from point to non-point sources (diffuse pollution) since the late 1960s (Heathwaite et al., 2000; Langan et al., 1997) and recent research focuses on the generation, transport and impact of diffuse pollutants (Ferrier et al., 2005). Other freshwater pollutants include lead (Farmer et al., 1997), silicon (Neal et al., 2005), pesticides (Harper et al., 1977) and suspended solids (Ferrier et al., 2001). The critical load concept has also been applied to freshwater ecosystems (Battarbee et al., 1996; Kernan et al., 2001)

Nitrogen and other pollutants also affect marine ecosystems (Wade et al., 2002). Untreated sewage is a declining source of pollution in Europe but anti-fouling paint and other sources of pollution continue to cause concern (Matthiessen & Law, 2002) and some pharmaceutical compounds remain persistent despite wastewater treatment (Roberts & Thomas, 2006). Heavy-metals affect marine life (Daka & Hawkins, 2006) but their concentrations in sediments have declined over the last 30 years (Duquesne et al., 2006). Oil spills remain a major and very visible threat to marine biodiversity (Law & Kelly, 2004; Long & Holdway, 2002; Velando et al., 2005). Exploitation and habitat loss are, however, seen as more important drivers of biodiversity loss in marine ecosystems (Dulvy et al., 2003). Nevertheless, long-term studies of the impact of polluted areas are very uncommon (Hawkins et al., 2002).

H. Inclusion of biodiversity aspects into flood risk management plans

Flood risk management and river restoration have been given a new impetus from the Water Framework and Habitats Directives (Newson & Large, 2006) and it is becoming increasingly apparent that flood risk management and biodiversity are closely linked. Overgrazing, for example, has a negative impact on both flood risk and biodiversity (Sansom, 1999). Tree shelter belts may influence flood risk, providing opportunities for the conservation and restoration of biodiversity (Carroll et al., 2004).

I. Development of cohesion and structural funds towards directly of indirectly benefiting biodiversity

There is no research in this area in the UK as far as the authors are aware.

J. Participation of civil society in Environmental Impact Assessment (EIA) and other processes

There is increasing participation of civil society in the management of marine protected areas (Jones & Burgess, 2005), monitoring of biodiversity (Danielsen et al., 2005), and other conservation-related initiatives (Wood, 2005). Research on this topic has included studies on attitudes towards participation (Davos et al., 2002), the implications of local participation for national strategy (Gillingham & Lee, 1999)and its effectiveness (Goodwin, 1998; Goodwin, 1999). The latter has included research on participation in Environmental Impact Assessment (EIA) (Hartley & Wood, 2005; Snell & Cowell, 2006) and the use of multicriteria analysis in EIA (Balasubramaniam & Voulvoulis, 2005). There have also been studies on specific aspects of conservation (Twyman, 2000), stakeholder analysis on particular conflicts (Redpath et al., 2004) and conflicts generally (Young et al., 2005). The issue of farmers' involvement in agri-environment schemes is discussed above.

UK researchers have been active in research on or projects on the participation of local communities in initiatives related to the conservation and/or sustainable use of biodiversity or EIA in, for example, Ghana (Wiggins et al., 2004), Nepal (Bajracharya et al., 2006), Nicaragua (Hawkesworth & Perez, 2003), Cameroon (Malleson, 2002; Sharpe, 1998), Botswana (Davies, 2001), Belize (Oreszczyn &

Lane, 2000), Tanzania (Few, 2000) Nigeria and the Niger Delta (Adomokai & Sheate, 2004).

K. Ecological coherence and functioning and its inclusion into spatial planning

This is currently a particularly active area for research. Several national agencies have research programmes focusing on ecological connectivity, and developing tools to support planning decisions, e.g. Forest Research's habitat network programme (Watts et al., 2005b), collaboration with the Countryside Council for Wales (Watts et al, 2005a) to develop a woodland habitat network for Wales in support of the Wales Woodland Strategy; and work within Natural England (Catchpole, 2006).

The Bioforum project included a major consideration of ecological (social and economic) aspects relevant to spatial planning (Nowicki *et al.*, 2005). The placement of coastal defence structures, for example, can have a profound impact on biodiversity (Airoldi *et al.*, 2005).

3. CONCLUSIONS

Despite the research done on the topics covered by this review, much more research is needed. We need to know much more, in particular,, to be able to make accurate predictions. Most research (although not all) is small scale and limited in the factors it considers. We need much more broad based, process oriented research so that we understand whole systems. Very little research undertaken incorporates social and economic analysis. Land managers respond to a wide variety of factors in coming to their decisions. Our research needs to reflect this integrated, interdisciplinary context.

Despite the breadth of this review and the number of studies referred to, this report should be considered as a partial assessment of the subject area. There are many more research topics under the subject of sustainable use of biodiversity than covered by the remit of this review.

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