MANAGING AGRICULTURAL LANDSCAPES FOR PRODUCTION OF MULTIPLE SERVICES: THE POLICY CHALLENGE

JEL classification: Q1, Q5

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Abstract. *There is increasing recognition that there are a range of environmental goods and services that are important to society as a whole, but may have little or no value to individual landowners whose land may contribute to the overall production of these services within the landscape. Many of these goods and services may be a minor output of any one parcel of land, but when aggregated across a landscape become important generally. Management of agricultural landscapes has typically been considered as an emergent property arising from the individual decisions of individual landowners. However, this leads to the potential for a "tyranny of small decisions" (Odum, 1982) that in aggregate can contribute to the erosion of the environmental commons. This paper outlines the evidence for landscape effects on ecological systems, and suggests that such systems should be managed at a scale greater than the farm. This in turn implies that agri-environment schemes can function with greater impact if implemented across landscapes, allowing efficiency gains required within the "sustainable intensification" agenda. The challenge then is to derive policy instruments that can drive "top down" or "bottom up" implementation of such schemes such that neighbouring landowners do the "right thing" in the "right place". The proposed mechanism for "greening" the EU's Common Agricultural Policy is currently under debate: the extent to which the proposals are consistent with the overall need to balance biodiversity and production needs is discussed.*

Keywords: *Agri-environment, Ecosystem services, landscape management, common agricultural policy, sustainable agriculture*

1. Preamble: setting the scene

The global demand for food is undoubtedly increasing given both the growth in global population and the change in demand for food as wealth increases (Godfray *et al.*, 2010; Foresight, 2011; Tilman *et al.*, 2011a), particularly evident in developing countries with the nutritional transition from predominantly vegetarian diets to one with a greater meat and dairy component.

Simultaneously with demand growth, there are two main inhibitory drivers: competition for land and climate change. There is globally limited scope for expansion of agricultural land; and the majority of recent expansion of agricultural land has come at the expense of tropical forest (Lambin and Meyfroidt, 2011; Tilman *et al.*, 2011a) – with the ensuing societal costs of loss of natural capital and emission of greenhouse gases with the resulting considerable mitigation costs

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levied (TEEB, 2010). In addition, urbanisation of the population is changing the relationship between society and the land, not least as rural populations are often decreasing, reducing access to labour capital and transport and leading to changes in agricultural practice. Land is also increasingly used for non-food crops such as oil palm and environmental degradation has also led to abandonment of former agricultural land (Jobin *et al.*, 2001; Holmgren, 2006) Climate change is also likely to have major impacts on agricultural productivity and practices (Lobell *et al.*, 2008; Battisti and Naylor, 2009); a recent study suggests that on average by 2050 yields in sub-Saharan African agriculture will decrease between 7 and 27%, with higher productivity areas being more directly affected (Schlenker and Lobell, 2010).

There are therefore strong drivers underpinning the productionist agenda: increasing demand against the constraints of no more (perhaps less) land and climate change. However, there is also increasing recognition that the environment provides an important range of services ("ecosystem services") that need protecting (MEA, 2005; NEA, 2011). These include those that may aid food production (such as soil fertility, pollination, natural pest control, water) or may have monetary or non-monetary value for society as a whole (e.g. contributing to climate control by storing carbon, flood control, cultural such as the look of the landscape and the existence of iconic biodiversity) (MEA, 2005; NEA, 2011). The history of recent decades suggests that the green revolution has often come with unsustainable environmental impacts and resource use in terms of inorganic nitrogen, phosphate fertiliser, fuel use, soil degradation and also biodiversity loss, with the resulting degradation of ecosystem services upon which both agricultural productivity (in the long term) and society relies (Chamberlain *et al.*, 2000; Benton *et al.*, 2002; Robinson and Sutherland, 2002; Foresight, 2011; NEA, 2011). Thus, there is a third constraint acting against the productionist agenda: the need to reduce the environmental impact of agricultural practices, and increase its sustainability. This "sustainability challenge" is a very real one, because in the long run, sustaining production requires it and, as with climate change (Stern, 2007), the longer that agriculture fails to embrace the agenda, and develop the business opportunities it brings, the more, in the long run it is likely to cost to get back on track (both in monetary values, ecosystem losses and human costs).

The context outlined above leads to the notion of "sustainable intensification": producing more food per unit area, with fewer inputs, whilst minimising or mitigating environmental costs (Baulcombe, 2010; Foresight, 2011; Tilman *et al.*, 2011b). The question is: how to do it? There are two broad approaches leading to sustainable intensification: (a) promoting greater resource use efficiency, and (b) management of non-production areas within the farmed landscape to support ecosystem services.

Increasing efficiency within intensive agriculture is underpinned by a range of modern approaches (Gebbers and Adamchuk, 2010). These include tillage practices (low or no-till) and ensuring that agro-chemicals are used according to the "4Rs" principal (Mikkelsen, 2011): right intervention, right amount, right time, right place. Much of the potential of precision agriculture is yet to be realised, as technology is not yet fully developed; however, one can see a future fully embracing modern sensing (in-field sensors and remote sensing), self-guided robotic machinery capable of delivering site-specific management on the scale of single plants, and where livestock husbandry (e.g. diets and medicinal interventions) are tailored to individual animal's particular phenotypic needs (Wathes *et al.*, 2008). Furthermore, the potential for wastes to be a resource is increasingly being recognised (e.g. for organic fertiliser or for anerobic digestion to provide energy for the farms' needs). Thus, in terms of within-field management, agriculture is already on a journey to increase its efficiency and grow, or maintain, yields whilst using fewer resources. Such advances in resource use efficiency have a range of benefits (e.g. the economic benefits of farmers using less chemical and maximising its utility, and with a reduction in chemical use, a reduction in indirect environmental effects).

However, resource use efficiency is necessary, but not sufficient, to deliver the totality of the sustainable intensification agenda. In particular, a range of ecosystem services are derived from organisms living within the farmed landscape but which cannot exist (solely or in part) within a modern field. Provision of habitat is required to support beneficial animals (like bees, hoverflies and other pollinators, and small wasps and flies, which provide natural pest control) as well as animals and plants of cultural importance (e.g. butterflies, birds, mammals, meadow flowers). Furthermore, wooded non-production habitat can supplies a range of additional services including providing habitat, timber, carbon storage and influencing the water cycle by affecting rainfall, as transpiration may contribute to cloud formation in some parts of the world (Garcia-Carreras *et al.*, 2010). Non-production areas are also used as an intervention to reduce run-off improving water quality (Stutter *et al.*, 2012). They also directly impact on the "look" of the landscape, and directly contribute to its cultural value through this route (NEA, 2011). Many of the ecosystem services provided by the non-production land has none or marginal economic benefit for the local landowner, but considerable importance for society as a whole. Thus, similarly to the tragedy of the commons, if the societal benefits accruing from the non-production areas are not recognised and such areas not actively managed, then there is a risk to the service provision as a whole. Maintaining functioning ecosystem services requires concerted actions across large areas. Ignoring the needs of the birds and the bees on a single farm is unlikely to impact on their overall population sizes in a landscape because population processes occur at a scale larger than an individual farm. However, if every farmer ignores their needs, the "tyranny of small decisions" (Odum, 1982) leads to the large-scale erosion of their habitat, their populations and the services they provide.

2. The ecology of a farm depends on the surrounding landscape

Recent agroecological work emphasises that there are strong influences of the landscape[1] on the ecosystem within a farm or field (Bengtsson *et al.*, 2005; Gabriel *et al.*, 2006; Carre *et al.*, 2009; Chamberlain *et al.*, 2010; Diekotter *et al.*, 2010; Gabriel *et al.*, 2010; Batary *et al.*, 2011). This landscape effect arises because the range of available habitats coupled with the regional biota determines the local pool of species that can colonise an individual farm. Furthermore, individuals may require several habitats (e.g. overwintering, nesting and foraging habitats), and some species may move quite widely over their lifetimes: so whether individuals are observed in one place may depend on suitable habitat provision both in that place and at some greater distance. The organisms that a farmer might find on his or her land will therefore not solely depend on his management practices, but will also depend on the state of the environment in the surrounding landscape. As the surrounding landscape is a mixture of agricultural land and non-agricultural land, a farm's biodiversity also depends in part on the way neighbouring farms are managed (Gabriel *et al.*, 2010; Sutherland *et al.*, 2011).

To illustrate neighbourhood effects in more detail: organic farming causes on-farm increases in biodiversity. The extent of this increase varies with the locality but averages about 12% when

[1] where landscape is an arbitrary geographical area containing many farms

many groups are considered. If organic farms sit in neighbourhoods where organic farms are common, the average biodiversity is almost double at 20%, this increase being caused solely by neighbourhood farming practice (Gabriel *et al.*, 2010). If one considers that organic conversion is often more likely in landscapes that may be naturally higher in biodiversity because the locality may have constraints on high productivity, such as topology or climate (Gabriel *et al.*, 2009), then landscapes with many organic farms are often considerably more biodiverse than landscapes without (Bengtsson *et al.*, 2005; Hole *et al.*, 2005). However, organic farms that are very isolated in a highly productive region, perhaps where there is little natural biodiversity to colonise the farms, and where the farms alone have insufficient area to support long-term populations, may have biodiversity that is little different from a conventional farm (Brittain *et al.*, 2010; Gabriel *et al.*, 2010). Indeed, controlling for a range of landscape factors other than neighbourhood management indicates that a conventional farm in a landscape with organic farms in is typically no different from an organic farm in a conventional landscape in terms of biodiversity (Gabriel *et al.*, 2010).

Just as the benefits to wildlife of an action depend on the action, the neighbourhood management and the landscape, so do the costs (in terms of lost production arising from changing agricultural practices). If a farm is in an area where there are naturally small fields, valleys and climate that is not conducive to large-scale arable production, it may both be relatively unproductive in terms of yields and exist in wildlife-friendly surroundings (due to the many small areas of non-production land). Thus there may be relatively small differences in total yield between intensive and extensive farming (the extensive farmer may, in fact, gain in economic terms by producing a premium product), and at the same time there may be marked wildlife benefits by promoting extensive landscapes. Conversely, in a high production landscape, the total yield of extensive farms may be much less than intensive farming (which may not be compensated economically by premium production if the yield reduction is large), and at the same time, extensive farms may also not gain much in terms of biodiversity due to smaller local species pool (Hodgson *et al.*, 2010).

In addition to neighbouring farming practice interacting with the landscape to influence on-farm ecology, neighbourhoods also matter in socio-economic terms: where there are a critical mass of farmers doing the same thing in an area, a market may develop and, if farmers benefit from ecosystem services like pollination or natural pest control, they can gain from the increased ecosystem services that eventuates from landscape-level habitat availability (Sutherland *et al.*, 2011).

3. Ecosystem services should be managed at the landscape scale

Many ecosystem services (e.g. pollination, natural pest control, the maintenance of culturally important biodiversity) require non-production habitat. The smaller an individual patch of habitat, the fewer organisms can survive upon it. The more isolated a patch of habitat is, the more likely a small population will go extinct (e.g. by not finding any mates). A fundamental tenet of ecology is that for a population to persist, it needs habitat that is either a sufficiently large block, allowing a large population to exist, or if it is fragmented, that the fragments are close enough for ecological connectivity – i.e. for organisms to move from one patch to another via dispersal. Clearly, different organisms have different requirements both in terms of habitat type (such as areas of grass, hedges, woodland, water courses) and also in terms of the distance required for patches to be connected ecologically (Weibull *et al.*, 2003). An agricultural landscape with a diversity of habitat patches scattered across it, connected by a range of linear features (e.g. field margins) therefore supports high biodiversity (Benton *et al.*, 2003). The requirement to deliver landscape-wide habitat to enhance ecosystem services in agricultural areas has been a recent focus of the literature (Benton *et al.*, 2003; Tscharntke *et al.*, 2005; Fischer *et al.*, 2006; Ricketts *et al.*, 2008; Nelson *et al.*, 2009; Gabriel *et al.*, 2010; Benton *et al.*, 2011; Boughey *et al.*, 2011).

A landscape-level network of ecological areas is not inconsistent with sustainable farm businesses for two reasons. Firstly, on most farms there are a range of landscape elements which are non-cropped areas (e.g. field margins, hedgerows, ditches). With the increasing use of precision farming techniques (such as yield monitoring) farmers also may often identify areas which are uneconomical to farm (due to local soil, drainage or access constraints). Secondly, farmers may benefit directly (in production terms) or indirectly via maintaining some non-cropped areas for wildlife. For example, the beneficial insects (such as pollinators and natural pest control agents) typically require non-cropped habitat for nesting and over-wintering. Thus, if farmers maintain grassy margins, they may reduce the incidence of aphid outbreak in the adjacent arable fields, and therefore require less plant protection products. Similarly, marginal strips may act as cover crops for shooting purposes, buffer strips for preventing soil erosion and run-off of synthetic nitrogen (and the potential regulatory costs imposed). There are also potential positive impacts from non-disturbed ground into fields that may influence soil communities and soil fertility (Manning *et al.*, 2006; Ramette and Tiedje, 2007). Non cropped areas can potential provide other services, such as production of domestic fuel in terms of timber, and for many farmers, there are non-economic gains that can arise from public perception of the positive impacts of stewardship on the countryside (NEA, 2011).

Thus, an agricultural landscape that is a mixture of farmed land and a network of non-cropped areas of various types can potentially provide both the agricultural business that farmers rely on as well as contribute to the common societal goods of protecting and enhancing biodiversity and the services it provides. A landscape that is farmed extensively (say organically) does not necessarily have greater potential for biodiversity and ecosystem service provision than a landscape that is farmed intensively, as long as the land not managed for agricultural production is actively and appropriately managed to provide a diverse network of non-cropped areas.

4. Policy challenges

The discussion above lays the case for recognising that sustainable agricultural landscapes require both within-field resource use efficiency and also a network of non-production habitat, suitable to the overall location. This creates a series of challenges to implement agri-enviornmental schemes as it implies that (a) what is “best” to do to optimise ecosystem services in production landscapes will vary by location, (b) that the same actions by landowners in different locations can have different impacts suggesting site-specific incentives, and (c) there are benefits to coordinating actions across landholdings, scaling up from single farms to landscapes.

a) The challenges of location-specific actions

Landscape factors ensure contrasting benefits of the same intervention in different locations, suggesting that tailoring actions to locations would be beneficial. This location-specific requirement will provide a challenge for policy formulation and implementation at a very large scale. One can imagine a common policy framework, setting the overall aims and process for making decisions, and the evidence base required to inform the decisions, with implementation devolved down to an administrative level at a granularity appropriate to the granularity of the landscape (e.g. a county level, or a regional level). Such a process would be considerably more expensive to implement than a more uniform policy process, but the gains in the resulting ecosystem services (if assigned any reasonable monetary value (NEA, 2011)) would make this investment worthwhile.

It is currently possible, within a research framework, to develop models that can predict how ecosystem service provision can vary with the landscape configuration (i.e. the network of patch sizes, types and distances between, and the matrix of production land). These models can also predict farm yields. They therefore allow the exploration of the joint relationship between farm outputs and ecosystem service provision, and how this relates to the configuration of patches of non-cropped land. Using such models one can specify the configuration that best optimises farm yields and landscape level services (Nelson *et al.*, 2009). Thus, at least in principle, each individual landscape can be optimised to maximise delivery of ecosystem services given the need to ensure farmers' economic returns.

b) The same action gets place-specific rewards

An often expressed principle of equity is that someone doing the same job should gain the same rewards as another(Fawcett, 1918), so there is a natural reluctance to reward the same action differentially simply depending on location. However, if we consider changing the conceptual framework from supporting the action to supporting the outcomes – "payment for ecosystem services" (Jack *et al.*, 2008) - then it becomes less contentious. Farm managers, undertaking identical management in different locations, naturally understand that yields will vary according to local factors; so if standard farm-management gains place-specific rewards, it is not entirely evident that agri-environment management should not do so too.

c) Coordination between farmers

The benefits of scaling up from the farm to landscape scale in agri-environment management have been highlighted before (Gabriel *et al.*, 2010). The challenge of agri-environment implementation is to find ways to encourage neighbouring farmers to do similar things, such that benefits arise at the landscape scale. One can imagine both top down and bottom up approaches. The top down approach would be to set an incentive scheme that preferentially rewards the options most beneficial to the locality, in the expectation that neighbours will make similar choices. The bottom up approach would be preferentially to reward neighbourhood cooperation, such that farms can develop cooperative ideas and receive preferential rewards for the more beneficial approach. To illustrate this, imagine if, in a specific location, it is appropriate to maximise the area of a block under nature management (say as nesting habitat for an iconic bird). Three neighbouring farmers could add considerable value to any land they set aside for nature if they set land aside at the intersection of their boundaries thereby creating a single large block (Hodgson *et al.*, 2010). This should be preferentially rewarded relative to them creating three separated blocks.

5. The CAP proposals as a case study of modern agri-environment management

In October 2011 the European Commission forwarded proposals for reform of the Common Agricultural Policy (hereafter CAP) to the European Council and Parliament, initiating discussions that will last up to mid-2013. The proposals aim to generate a compulsory amount of non-cropped "ecological focus areas" (EFAs) comprising 7% of the total farmed area across the landscape. This has potential benefit in terms of being a mechanism to provide a landscape scale network. The UK Curry report first called for "broad and shallow" interventions to produce more sustainable landscapes in 2002 (Curry, 2002), and ensuring that all farms manage EFAs is consistent with this approach. The proposals also aim to encourage landscape heterogeneity by stimulating production of multiple crops (including permanent pasture). Encouraging greater heterogeneity has also been widely discussed for a decade (Benton *et al.*, 2003). Thus, at first sight, the CAP proposals, in principle, could be beneficial.

The "ideal" CAP reforms would be to encourage landscape measures that used the EFAs to create a "well connected" ecological landscape (with patches of suitable habitat sufficiently close for dispersal/movement from patch to patch), with a range of linear habitat types (e.g. flower rich margins, grassy margins, hedgerows) creating the connectivity and with patches of land for non-agricultural specialists (e.g. woods, grasslands, wetlands). Furthermore, as outlined above, the specificities of the "ideal landscape configuration" will vary from location to location, depending on the local habitat, geography and biota. Thus, the considerations for landscape-scale management of habitat: (1) the amount and type of habitat, (2) its spatial extent, (3) its spatial distribution and the connectivity it brings to a landscape, (4) the quality of the habitat created and, (5) its location-specificity. The current proposals address (1) (by specifying a percentage of EFAs) and (2) by making receipt of the Pillar 1 payment contingent on this, ensuring broad take-up of the prescription across very large areas.

The degree to which the proposals really contribute positively to environmental outcomes will depend on the final details. If the EFAs are left as unmanaged set aside, the areas will have rather less ecological value than if they were actively managed to provide good habitat (Sotherton, 1998) and therefore, overall, the greening proposals would have little benefit (relative to the potential cost of removing productive land from production). Conversely, if the land is actively managed to ensure delivery of ecosystem services a landscape with a 7% network of "green veins" could be highly beneficial. Thus, the requirement for active management is key to whether the greening proposals will create a net benefit. Payment for service delivery would allow land managers to undertake what they are interested in doing, with perhaps farmers having the greatest interest in delivering the maximum gains, leading to landscape appropriate actions. Discussion of how to get the greatest gains would lead to productive partnerships between farmers and agri-environment advisors in agencies or NGOs.

After management, the second most important issue is the spatial layout of the EFAs. A connected landscape requires that there is sufficient connection between habitats for a range of organisms to live, and disperse along/between linear features or patches. Thus, ensuring there is a mix of features, patch sizes and habitats is crucial to developing a connected landscape (7% as a single block will not be as good as 7% as a network for the majority of species). This implies that for significant gains to be made in delivery of ecosystem services, connected landscapes should be planned and advisors work with farmers to guide their actions towards those that will be most beneficial. To a certain extent, at the farm level, this already occurs within some schemes (e.g. The UK's Higher Level Schemes, Australia's Box Gum Grassy Woodland Environmental Stewardship Scheme). Furthermore, owing to the location-specific costs and benefits of interventions, the impact of 7% of land set aside into EFAs will vary: in some places the optimised landscape to deliver production and ecosystem services will require more or less than this figure. This implies that allowing member states the potential to vary this figure locally (e.g. by balancing Pillar 1 payments with Pillar 2 agri-environment payments), whilst maintaining an average 7%, would be beneficial.

Encouraging spatial heterogeneity to promote biodiversity gains is a sensible option as identified frequently in the ecology literature. The proposals aim to promote this by encouraging "arable land to consist of 3 different crops simultaneously". However, to make this meaningful ecologically, the heterogeneity needs to be promoted on an absolute spatial scale (e.g. a 2 x 2 km landscape) not a relative scale (the farm). To illustrate this, a very large farm could comply but still maintain areas of homogeneity greater than the total area of a small farm.

6. Conclusions

Sustainable intensification is necessary if the issue of looming global food insecurity is to be avoided (Foresight, 2011). Environmental sustainability is necessary, by definition, to sustain agricultural production into the long term. Sustainability has many elements that contribute at different spatial scales, from very local actions on soil to improve conditions in a particular small patch, to the farm contributing to a landscape of habitat providing a range of ecosystem services that the land manager may not get value from, but is valuable for society (e.g. biodiversity conservation, carbon storage, water quality). As has long been recognised in the literature on water resource management, the appropriate scale to consider ecosystem service provision is a larger scale than the scale of agricultural management (Van Zyl, 1995; Pollard and Huxham, 1998; Fenemor *et al.*, 2011). For water, there is a natural scale, that of the "catchment"; for other ecosystem services, there is no natural scale that applies across all services, but the appropriate scale is greater than the farm and is one that can reflect natural variation in underlying climate, topography etc. This landscape scale would be at a scale of 10s to 100s of square kilometres.

Large scale, integrated management, is possible: the EU's Water Framework Directive is large scale management, judged by compliance with quality standards, and requires site-specific assessment and actions to produce set outcomes. Agricultural landscapes need to be approached in a way similar to that in which catchments are managed for water quality. If this happens, sustainable intensification of agricultural landscapes will be possible, without further eroding the ecosystem services that require non-production land.

REFERENCES

Batary, P., Andras, B., Kleijn, D., Tscharntke, T., 2011. Landscape-moderated biodiversity effects of agri-environmental management: a meta-analysis. Proc. R. Soc. Lond., Ser. B: Biol. Sci. 278, 1894-1902.

Battisti, D.S., Naylor, R.L., 2009. Historical warningsof future food insecurity with unprecedented seasonal heat. Science 323, 240-244.

Baulcombe, D., 2010. Reaping benefits of crop research. Science 327, 761.

Bengtsson, J., Ahnstrom, J., Weibull, A.C., 2005. The effects of organic agriculture on biodiversity and abundance: a meta-analysis. J. Appl. Ecol. 42, 261-269.

Benton, T.G., Bryant, D.M., Cole, L., Crick, H.Q.P., 2002. Linking agricultural practice to insect and bird populations: a historical study over three decades. J. Appl. Ecol. 39, 673-687.

Benton, T.G., Dougill, A.J., Fraser, E.D.G., Howlett, D.J.B., 2011. The scale for managing production vs the scale required for ecosystem service production. World Agriculture 2, 11.

Benton, T.G., Vickery, J.A., Wilson, J.D., 2003. Farmland biodiversity: is habitat heterogeneity the key? Trends Ecol. Evol. 18, 182-188.

Boughey, K.L., Lake, I.R., Haysom, K.A., Dolman, P.M., 2011. Effects of landscape-scale broadleaved woodland configuration and extent on roost location for six bat species across the UK. Biol. Conserv. 144, 2300-2310.

Brittain, C., Bommarco, R., Vighi, M., Settele, J., Potts, S.G., 2010. Organic farming in isolated landscapes does not benefit flower-visiting insects and pollination. Biol. Conserv. 143, 1860-1867.

Carre, G., Roche, P., Chifflet, R., Morison, N., Bommarco, R., Harrison-Cripps, J., Krewenka, K., Potts, S.G., Roberts, S.P.M., Rodet, G., Settele, J., Steffan-Dewenter, I., Szentgyorgyi, H., Tscheulin, T., Westphal, C., Woyciechowski, M., Vaissiere, B.E., 2009. Landscape context and habitat type as drivers of bee diversity in European annual crops. Agric. Ecosyst. Env. 133, 40-47.

Chamberlain, D.E., Fuller, R.J., Bunce, R.G.H., Duckworth, J.C., Shrubb, M., 2000. Changes in the abundance of farmland birds in relation to the timing of agricultural intensification in England and Wales. J. Appl. Ecol. 37, 771-788.

Chamberlain, D.E., Joys, A., Johnson, P.J., Norton, L., Feber, R.E., Fuller, R.J., 2010. Does organic farming benefit farmland birds in winter? Biol. Lett. 6, 82-84.

Curry, D., 2002. Farming and food: a sustainable future., Policy Commission Report. DEFRA, London.

Diekotter, T., Wamser, S., Wolters, V., Birkhofer, K., 2010. Landscape and management effects on structure and function of soil arthropod communities in winter wheat. Agric., Ecosyst. Environ. 137, 108-112.

Fawcett, M.G., 1918. Equal pay for equal work. The Economic Journal 28, 1-6.

Fenemor, A., Phillips, C., Allen, W., Young, R., Harmsworth, G., Bowden, B., Basher, L., Gillespie, P., Kilvington, M., Davies-Colley, R., 2011. Integrated catchment management—interweaving social process and science knowledge. N. Z. J. Mar. Freshwat. Res. 45, 313-331.

Fischer, J., Lindenmayer, D.B., Manning, A.D., 2006. Biodiversity, ecosystem function, and resilience: ten guiding principles for commodity production landscapes. Front. Ecol. Environ. 4, 80-86.

Foresight, 2011. The Future of Food and Farming: challenges and choices for global sustainability. The Government Office for Science, London.

Gabriel, D., Carver, S.J., Durham, H., Kunin, W.E., Palmer, R.C., Sait, S.M., Stagl, S., Benton, T.G., 2009. The spatial aggregation of organic farming in England and its underlying environmental correlates. J. Appl. Ecol. 46, 323-333.

Gabriel, D., Roschewitz, I., Tscharntke, T., Thies, C., 2006. Beta diversity at different spatial scales: Plant communities in organic and conventional agriculture. Ecol. Appl. 16, 2011-2021.

Gabriel, D., Sait, S.M., Hodgson, J.A., Schmutz, U., Kunin, W.E., Benton, T.G., 2010. Scale matters: the impact of organic farming on biodiversity at different spatial scales. Ecol. Lett. 13, 858-869.

Garcia-Carreras, L., Parker, D.J., Marsham, J.H., 2010. What is the mechanism for the modification of convective cloud distributionby land surface-induced flows? Journal of the Atmospheric Sciences 68, 619-634.

Gebbers, R., Adamchuk, V.I., 2010. Precision agriculture and food security. Science 327, 828-831.

Godfray, H.C.J., Beddington, J.R., Crute, I.R., Haddad, L., Lawrence, D., Muir, J.F., Pretty, J., Robinson, S., Thomas, S.M., Toulmin, C., 2010. Food Security: the Challenge of Feeding 9 Billion People. Science 327, 812-818.

Hodgson, J.A., Kunin, W.E., Thomas, C.D., Benton, T.G., Gabriel, D., 2010. Comparing organic farming and land sparing: optimizing yield and butterfly populations at a landscape scale. Ecol. Lett. 13, 1358-1367.

Hole, D.G., Perkins, A.J., Wilson, J.D., Alexander, I.H., Grice, P.V., Evans, A.D., 2005. Does organic farming benefit biodiversity? Biol. Conserv. 122, 113-130.

Holmgren, P., 2006. Global Land use area change matrix. Forest Resources Assesment Working Paper, Rome.

Jack, B.K., Kousky, C., Sims, K.R.E., 2008. Designing payments for ecosystem services: lessons from previous experience with incentive-based mechanisms. Proceedings of the National Academy of Sciences 105, 9465-9470.

Jobin, B., Choiniere, L., Belanger, L., 2001. Bird use of three types of field margins in relation to intensive agriculture in Quebec, Canada. Agric. Ecosyst. Env. 84, 131-143.

Lambin, E.F., Meyfroidt, P., 2011. Global land use change, economic globalization, and the looming land scarcity. Proceedings of the National Academy of Sciences of the United States of America 108, 3465-3472.

Lobell, D.B., Burke, M.B., Tebaldi, C., Mastrandrea, M.D., Falcon, W.P., Naylor, R.L., 2008. Prioritizing climate change adaptation needs for food security in 2030. Science 319, 607-610.

Manning, A.D., Fischer, J., Lindenmayer, D.B., 2006. Scattered trees are keystone structures - implications for conservation. Biol. Conserv. 132, 311-321.

MEA, 2005. Millennium ecosystem assessment. Ecosystems and Human Well-being: Synthesis Washington DC: Island Press.

Mikkelsen, R.L., 2011. The “4R” Nutrient Stewardship Framework for Horticulture. HortTechnology 21, 658-662.

NEA, U., 2011. The UK National Ecosystem Assessment. Synthesis of the Key Findings.

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D.R., Chan, K.M.A., Daily, G.C., Goldstein, J., Kareiva, P.M., Lonsdorf, E., Naidoo, R., Ricketts, T.H., Shaw, M.R., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Front. Ecol. Environ. 7, 4-11.

Odum, W.E., 1982. Environmental degradation and the tyranny of small decisions. Bioscience 32, 728-729.

Pollard, P., Huxham, M., 1998. The European Water Framework Directive: a new era in the management of aquatic ecosystem health? Aquat. Conserv.: Mar. Freshwat. Ecosyst. 8, 773-792.

Ramette, A., Tiedje, J.M., 2007. Multiscale responses of microbial life to spatial distance and environmental heterogeneity in a patchy ecosystem. Proceedings of the National Academy of Sciences of the United States of America 104, 2761.

Ricketts, T.H., Regetz, J., Steffan-Dewenter, I., Cunningham, S.A., Kremen, C., Bogdanski, A., Gemmill-Herren, B., Greenleaf, S.S., Klein, A.M., Mayfield, M.M., Morandin, L.A., Ochieng, A., Potts, S.G., Viana, B.F., 2008. Landscape effects on crop pollination services: are there general patterns? Ecol. Lett. 11, 499-515.

Robinson, R.A., Sutherland, W.J., 2002. Post-war changes in arable farming and biodiversity in Great Britain. J. Appl. Ecol. 39, 157-176.

Schlenker, W., Lobell, D.B., 2010. Robust negative impacts of climate change on African agriculture. Environmental Research Letters 5.

Sotherton, N.W., 1998. Land use changes and the decline of farmland wildlife: an appraisal of the set-aside approach. Biol. Conserv. 83, 259-268.

Stern, N.H., 2007. The economics of climate change: the Stern review. Cambridge Univ Pr.

Stutter, M.I., Chardon, W.J., Kronvang, B., 2012. Riparian Buffer Strips as a Multifunctional Management Tool in Agricultural Landscapes: Introduction. J. Environ. Qual. 41, 297-303.

Sutherland, L.A., Gabriel, D., Hathaway-Jenkins, L., Pascual, U., Schmutz, U., Rigby, D., Godwin, R., Sait, S.M., Sakrabani, R., Kunin, W.E., Benton, T.G., Stagl, S., 2011. The ‘Neighbourhood Effect’: a multidisciplinary assessment of the case for farmer co-ordination in agri-environmental programmes. Land Use Policy.

TEEB, 2010. Mainstreaming the Economics of Nature: a synthesis of the approach, conclusions and recommendations of TEEB.

Tilman, D., Balzer, C., Hill, J., Befort, B.L., 2011a. Global food demand and the sustainable intensification of agriculture. Proceedings of the National Academy of Sciences of the United States of America 108, 20260-20264.

Tilman, D., Balzer, C., Hill, J., Befort, B.L., 2011b. Global food demand and the sustainable intensification of agriculture. Proceedings of the National Academy of Sciences.

Tscharntke, T., Klein, A.M., Kruess, A., Steffan-Dewenter, I., Thies, C., 2005. Landscape perspectives on agricultural intensification and biodiversity - ecosystem service management. Ecol. Lett. 8, 857-874.

Van Zyl, F., 1995. Integrated catchment management: is it wishful thinking or can it succeed? Water Science and Technology 32, 27-35.

Wathes, C.M., Kristensen, H.H., Aerts, J.M., Berckmans, D., 2008. Is precision livestock farming an engineer's daydream or nightmare, an animal's friend or foe, and a farmer's panacea or pitfall? Comput. Electron. Agric. 64, 2-10.

Weibull, A.C., Ostman, O., Granqvist, A., 2003. Species richness in agroecosystems: the effect of landscape, habitat and farm management. Biodivers. Conserv. 12, 1335-1355.