Paper:
Firbank, LG, Elliott, J, Drake, B, Cao, Y and Gooday, R (2013) *Evidence of sustainable intensification among British farms.* Agriculture, Ecosystems and Environment, 173. 58 -65.

http://dx.doi.org/10.1016/j.agee.2013.04.010

Evidence of sustainable intensification among British farms

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ABSTRACT

Several influential reports have suggested that one of the most appropriate responses to expected food shortages and ongoing environmental degradation is sustainable intensification, i.e. the increase of food production with at worst no increase in environmental harm, and ideally environmental benefit. Here we sought evidence of sustainable intensification among British farmers by selecting innovative arable, dairy, mixed and upland farms and analysing their own data on yields, inputs and land use and management for 2006 and 2011. The evidence was obtained by interview, and was interpreted in terms of the ecosystem services of food production (GJ ha^{-1}, where area took into account estimated area to grow any imported animal feeds), regulation of climate, air and water quality (modelled

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emissions of GHGs (CO_2e ha), ammonia (kg ha) and nitrate loss (kg ha)) and biodiversity (using an index based on the presence of habitats and management).

Several farms have increased both food production and other ecosystem services over this time by increasing yields, using resources more efficiently and /or enhancing biodiversity, and sometimes by reducing livestock numbers and increasing cropping. The motivation has been to improve farm profitability through increasing food production, reducing input costs and accessing public payments through agri-environment schemes and generating renewable energy. Such sustainable intensification was not achieved by farmers who increased meat or milk yields.

Sustainable intensification can be achieved when the correct drivers are in place to influence the actions of individual farmers. Also, it is possible to indicate sustainable intensification by using a small number of high-level indicators derived from data that farmers already hold, though such an approach may not capture the impacts of farmer innovative practices.

HIGHLIGHTS

 Sustainable intensification is being achieved by innovative farmers in Britain Sustainable intensification is driven by the desire to raise income and cut costs Biodiversity enhancement is seen by farmers as a cost, to be borne by public payments Sustainable intensification can be indicated using farmers‟ own data but the metrics need to be refined Research is needed to capture the impacts of innovations such as zero tillage

KEYWORDS:

Ecosystem services, biodiversity, GHG emissions, diffuse pollution, ammonia emissions, land sparing, agro-ecological indicators, sustainable agriculture, organic farming

Introduction

During the second half of the 20th century, global agricultural production increased at rates that were sufficient to keep pace with demands. Concerns about global food security focused on issues of equity and distribution, rather than worries about the total amount of food available around the world (McIntyre et al., 2009). The global spike in food prices in 2007-08 changed perceptions markedly, and brought attention to the fact that global demand for food was starting to rise faster than supply. The concept of „sustainable intensification", in which "yields are increased without adverse environmental impact and without the cultivation of more land", was developed to highlight the need to improve agricultural productivity without incurring the kind of environmental costs associated with intensive agriculture in the past (Royal Society, 2009; see also Foresight, 2011). Some authors consider that sustainable intensification should go further than requiring no additional environmental harm, but should involve increases in both food production and the flow of ecosystem services (e.g. Firbank, 2009; Pretty et al., 2011).

While there are many cases in Africa where increases in yield have been associated with improved environmental outcomes, such as by improved management of highly degraded soils (Pretty et al., 2011), there is very little evidence of sustainable intensification among commercial farms of temperate regions. This is partly because most published studies of changing levels of production and environmental impacts have looked at interactions between two variables at the national level, rather than at the individual farm (Firbank et al., 2011), and partly because sustainable intensification is easier to observe from low baseline yields and environmental performance. Also, in temperate regions, sustainable intensification is widely perceived more as a strategy for the future than a desirable change in the present (Foresight, 2011). Yet the pressures on agricultural production in temperate regions are already increasing; at the time of writing, there is likely to be a drawdown of global cereal stocks because of the poor weather, including the prolonged and severe drought in the US (FAO, 2012), which itself is consistent with a shift towards less favourable weather conditions for temperate agriculture (Francis and Vavrus, 2012). It is therefore important to ask whether sustainable intensification is already being delivered by some farmers, what strategies they are adopting and why, what barriers they are facing and what are the inherent risks. Only once these questions are answered will it be possible to design interventions that will encourage sustainable intensification in appropriate situations.

Innovation in sustainable intensification is most likely to be found among the more progressive farms. Therefore, our approach was to identify a group of such farms, to test whether sustainable intensification has been achieved. We collected data on agricultural production and measures of environmental impact, and discussed with farmers their drivers, motives and perceived barriers to implementation of sustainable intensification. We relied entirely upon data already available to the farmer, so that the methodology can readily be applied to much larger samples of farms at low cost in due course, for example to support certification of environmental standards at the farm scale.

2 Methods

2.1 Quantifying sustainable intensification

Sustainable intensification is a process, rather than a condition at any time. In the absence of an agreed set of metrics of sustainable intensification, we adopted a very pragmatic approach. A farm was considered to be practicing sustainable intensification if food production per unit area had increased during the study period, and that none of the environmental variables had deteriorated. Changes were analysed in both agricultural production and a representative set of environmental variables from a baseline of 2005-6, i.e. before the global increases in food prices, to 2010-11, the most recent data for which most farms have data (in some cases, we had to use data from different years). We adopted a small number of variables, to allow a qualitative assessment of sustainable intensification (following Pretty (2008a,b)), which we regarded as being more transparent than interpreting sustainability by integrating a larger number of variables into a common unit, for example money (Bateman et al., 2011). The system boundary was the farm gate.

The measurables were taken from five major categories of ecosystem services that are known to have changed significantly across the UK on farmland (UKNEA, 2011), namely agricultural production, biodiversity, climate regulation, regulation of air quality and regulation of water quality. For ease of interpretation, we used a single variable to represent each of these categories for each individual farms, measured on the basis of land area, and per unit food production. We used only data already held by the farmers, interpreted using commentaries obtained during farm visits and interviews; this restriction excluded some ecosystem services and processes that could not therefore be reported with acceptable precision, notably landscape quality and levels of soil erosion from the farm.

Data were collected on food production by the whole farm area. This comprised the total land area of the farm (including non-productive areas), supplemented with estimates of the area of land required to produce feeds brought onto the farm. These estimates were obtained using data on generic compound formulation (Table 1) to break down compound feeds into estimated amounts of constituent crops (wheat, barley, oilseed rape), and then UK average yields of these crops were used to estimate areas of land used to grow them. We did not attempt to distinguish between different sources of such imports.

In order to generate a single measure of food production, we condensed the available data into gross energy per unit area, i.e. GJ ha^{-1}, where energy content of foodstuff were taken from multiple sources (Table 2) and the area was for the whole farm as calculated above. This metric allows changes to be tracked for an individual farm, but is not suitable for comparison across farms as it ignores variation in the financial and nutritional value of different foodstuffs and is closely related to the potential of the farmland for food production in terms of soil and climate. In order to avoid double counting, the energy content of cereals and fodder crops used for the farmers" own livestock were not included in the analyses.

Carbon footprints were estimated for the farms by combining two approaches. Carbon dioxide emissions (CO_2) were calculated using the CALM tool (CLA, 2012), that allows for energy use and land use change using values taken from UK Greenhouse Gas Emissions Inventory (DECC, 2012). Potential emissions of methane and nitrous oxide were calculated using the Farmscoper tool (Gooday and Anthony, 2010), that uses data on cropping, soil type and rainfall, livestock numbers, fertiliser use and housing to estimate gross emissions for each gas in kg ha^{-1}. Methane and nitrous oxide emissions were then converted into Global Warming Potential (in CO_2e) and combined with the CO_2 emissions to give a total carbon footprint. These estimates do not allow for variation in farm practices, for example how or when fertilizers are applied, and do not address carbon sequestration into the vegetation or soils.

Potential losses of ammonia to the atmosphere and nitrates to running waters were used as high level indicators of regulation of air and water quality. The Farmscoper tool (Gooday and Anthony, 2010) was used to calculate these losses taking into account soil, rainfall, the cropping and stocking and level of inputs. These values represent potential rather than actual losses, as the Farmscoper tool was not used to account for the use of technologies or farm practices used to mitigate against pollutant loss and does not cover some technologies such as precision farming.

Few of the farmers had their own biodiversity data. We therefore sought data on the protection goals supported by UK policy and enshrined in the UK Biodiversity Indicators, in terms of the habitats and resources available to wildlife, and the extent to which they were supported through agri-environment schemes or legal protection (Defra, 2011). Therefore we compiled scores for biodiversity „stock‟ and „change‟ from farmer responses to indicate the diversity and protection of habitats at the baseline, and the efforts being made to improve the farm for biodiversity since. The scoring also considered broader aspects of farm management known to have major impacts on farmland birds (Butler et al., 2007; Butler et al., 2009) and whether any set-aside arable land has been lost, which was an important habitat for birds and invertebrates (Firbank et al., 2003). While such scores are helpful when considering the potential impacts on biodiversity of changes on a farm or group of farms, they are not well related to actual numbers, diversity or conservation value of different taxa. In particular, a farm with a wide range of features would score more highly than a farm with a large area of a single, well managed, species-rich habitat. Therefore, to prevent over-interpretation of the data, the scores were simplified into 5 point scores for „biodiversity‟ stock (ranging from 1 to 5) and change (ranging from -2 to +2) (Table 3).

The comparisons between baselines and current indicator values for individual farms are subject to various levels of uncertainty, partly from the values given by farmers and models used to analyse them, and partly because of year-on-year variation in weather and market conditions. Therefore we are not able to ascribe formal levels of statistical significance to these data; instead, we have interpreted differences of +/-10% as not indicating change.

2.2 Selection of farmers

The 20 case study farmers were selected not at random, but according to whether they were considered to be innovative by their peers. Each farm had to be a mainstream, commercial operation in a single unit, not supported primarily from other sources of income, and not owned by a charity or research organisation. They were selected to represent the major farm types across Britain; 7 arable farms (A), 4 mixed (M), 4 dairy (D) and 5 upland livestock farms located in less favoured areas (LFA) (L) were chosen. Of these 20 farms, two were organic, 17 were in agri-environment schemes and 14 had installed, or plan to install, solar or wind power. One of the farms (A5) could not provide the required data for the analysis of carbon balance, and so is excluded from the GHG analyses.

Focus groups were also conducted to consider more broadly the practicalities of sustainable intensification, using the case study farms to stimulate discussion. Each focus group comprised eight or more farmers from a single sector, namely arable, dairy and LFA livestock, drawing on existing sector-based farmer discussion groups. The agenda of each group included capturing awareness and knowledge of the sustainable intensification concept, presentation of project aims and introduction of SI concept, strategies for achieving SI and discussion of strategies, indicators and challenges.

3 Results

3.1 Relationship between food production and environmental variables at baseline

There was no clear overall relationship between individual environmental variables and levels of food production across farm types at baseline (Fig 1), though the data hint at negative relationships between food production and emissions of nitrate and GHGs per unit area among dairy farms. Emissions of ammonia and GHGs were highest among dairy farms, and were similar among the other farm types. Nitrate losses per unit area were lowest among the upland livestock farms, and highest among two of the dairy farms, with much variation (Fig 1, Table 4). Baseline biodiversity scores varied within all farm sectors, with no obvious relationships with food production (Fig 1).

Food production per unit area varied greatly between farm types, ranging from a mean of 1.5 GJ ha^{-1} for the upland farms, 40 GJ ha^{-1} for dairy, 42 GJ ha^{-1} for mixed farming, and 101 GJ ha^{-1} for arable farms. These differences mean that the different farm types tend to be clustered in all graphs, with arable farms showing high levels of food production, LFA low levels and mixed and dairy farms intermediate levels.

3.2 Changes in food production

Of the 20 farms, 10 showed increased food production of over 10 % (Fig 2, Table 4). Some farmers increased levels of food production by increasing the productivity of the existing farming system. Thus farm A2 obtained a large increase in potato yield, attributed by the farmer to variety change; D2 increased yields of both milk and arable crops, and increased cattle numbers, while L3 increased numbers of cattle and sheep. Three of the four dairy farms increased milk production per unit area, with the greatest increase achieved by farm D2, which had been the most productive at baseline, while farm M3 achieved increases in the energy content of the food that was produced by reducing livestock numbers and increasing the area and yields of arable crops.

Two of the five farms that showed decreases in food production of over 10% (Fig 2, Table 4) were undergoing changes in farming system that reduced food output; farm L5 was in a period of transition to organic status, and had increased farm area but not livestock numbers, while A7 switched land away from cereals to fruit and vegetables, with lower energy contents and higher energy requirements. In all other cases, yield reductions were explained by the farmers in terms of annual variation of weather and cropping patterns.

3.3 Changes in other ecosystem services

The case study farms showed both improvement and deterioration of the non-food ecosystem services. The improvements in yields seen on the dairy farms were accompanied by worsening carbon footprint and nitrate and ammonia losses, with little change to biodiversity scores (Fig 3). These increased levels of pollution pressure per unit of land were not evident when indicators were calculated per unit of food production (Fig 4). Much greater variation was seen within the other farm types (Fig 3). Only farms A2, M3 and L1 showed substantial reductions in losses of all pollutants per unit food (Fig 4). The enhancement of biodiversity was not related to farm type or changes in food production (Fig 3).

3.4 Evidence of sustainable intensification

Three farms achieved sustainable intensification as defined within this study; two of the arable farms (A2 and A4) and one mixed farm, M3. The upland farm was very close to achieving sustainable intensification according to the definition used here, showing enhancements to all environmental variables and an increase in food production of 10% and (Table 3).

The interviews with the farmers revealed that Farm A2 deliberately sought to establish a sustainable business model, based around supply chain requirements and with the engagement of farm staff, through a combination of technology to improve resource efficiency, monitoring of performance and putting less productive land into grassland, buffers, hedgerows and woodland. The farmer has introduced technologies such as more fuel-efficient machinery, better insulation, and more efficient use of fertilisers. Farm A4 has used agri-environment funding to take poorer land out of production and enable easier and more cost-effective farming without impacting income. The farm has also benefitted from a 50 % reduction in fertiliser usage from variable rate applications, and from the installation of solar panels and reduced fuel use through minimum tillage. Farm M3 has changed a great deal since the baseline of 2006, with economic output doubled and investment in precision farming techniques, again using agri-environment schemes on the poorer land. The baseline figures for farm M3 were high for the sector, and the reductions in pollutants were driven partly by reductions in the livestock enterprise and partly by resource efficiencies such as the installation of two wind turbines and greater use of precision techniques. Farm L1 scored highly for food production and environmental quality by having reduced numbers of hill sheep in favour of a more prolific cross-bred ewe, improving grassland management to reduce inputs of both fertilisers and concentrates, and using agri-environment schemes to enhance the already diverse set of habitats.

Several farms increased food production at the expense of environmental quality per unit land. This was particularly noticeable among the dairy farms D1, D2 and D4, and the LFA farms L2 and L3, even though all of these farms have made investments to increase the efficiency of resource use. Levels of pollution per unit of food production were much more stable (Fig. 4). The arable farm A1 showed large increases in GHG emissions (Fig. 3), due to investments in new potato and vegetable stores and the running of new haulage lorries. The scores for this farm therefore included operations that were previously external to the farm, and may not represent an increase in GHG emissions across the whole food chain. Farm A3 showed increases in all pollutants per unit food (Fig. 4), largely due to the introduction of a chicken enterprise.

Only one farm (L5) increased environmental enhancement alongside a decline in food production (Table 3); this result was associated with conversion to organic status.

3.5 Results from the focus groups

Approximately half of all farmers in the focus groups were aware of the concept of SI; knowledge was highest in the arable sector and lowest in the dairy sector. In common with the case study farms, the primary underlying driver for farm business decisions was seen as financial sustainability. Farmers in all three focus groups favoured reducing inputs without affecting outputs, but there were some differences between sectors, for example the dairy group did not favour using poorer quality land for biodiversity as much as the others. The dairy focus group concluded that it may be harder to achieve sustainable intensification in this sector, not least because of the currently difficult market conditions. All focus groups considered that Single Farm Payments deflect from the drive for efficiency and adds to costs. There was also consensus on the need for access to R&D and help with investment in new technologies.

4 Discussion

This study shows that some British farm have been practising sustainable intensification, as defined within this study. At least three of the farms have increased food production whilst reducing pollution and enhancing biodiversity. They did so, not in response to calls for the sustainable intensification of global agriculture, but in response to the particular drivers and opportunities facing their businesses.

In general, farmers seek to determine their land use and land management to maximise their profits, subject to resource constraints and subject to factors beyond their control, including input costs (e.g. Rounsevell et al., (2003); Hanley et al., (2012)). Interviews with the farmers did not indicate that their business strategy had been strongly influenced by rising farm gate prices for many products. A greater concern was the desire to control the rising costs of inputs, which have risen sharply since 2006 (the financial value of fertiliser consumed in the UK has doubled, animal feed increased by 72 per cent and energy by 66 per cent (Defra et al., 2012)). Case study farmers have introduced technologies and strategies to reduce wastage of energy and nutrients and to manage risk though increasing the on-farm capacity to grow animal feed, store water, or to install renewable energy supplies. While such actions reduced the levels of pollution per unit area of several arable farms, no farm which increased animal production simultaneously reduced levels of potential pollution per unit area. This result is not surprising, as pollution levels from cattle and sheep are currently correlated with stocking densities (Cardenas et al., 2011) because they are closely related to enteric fermentation and manure management (Del Prado et al., 2011). Nevertheless, the models estimate potential pollution rather than actual pollution, because the models used do not account for mitigation options such as precision farming, nor the details of hydrology that influence how much potential pollution enters watercourses (see e.g. Hutchins et al. (2009)).

The quest to maximise profits also accounted for uptake of renewable energy generation and agri-environment schemes, even among farmers who expressed their commitment to sustainability. Fourteen of the twenty case studies used, or plan to use, renewable energy. Such investments provide an additional income stream through the UK Government‟s feed-in tariffs and reduce exposure to rising electricity prices; the reduction in carbon footprint was seen as a welcome additional benefit.

The case study farmers regarded the active management of land for biodiversity as a good thing to do, but also as a cost to be borne by external financial support. Seventeen of the farmers were in agri-environment schemes, seeking payments to enhance biodiversity on the less productive land. Such profit maximisation is a good predictor of the take up of agrienvironment schemes (e.g. Cary and Wilkinson (1997)), and confirms that some form of financial support and / or strong regulation is required to maintain population levels of many farmland taxa (e.g. Krebs et al. (1999); Baker et al. (2012); Doxa et al. (2012); Mouysset et al. (2012)), at least for the more innovative farmers. Several farmers mentioned they were adopting a land-sparing approach: this is an interesting example of how terminology can evolve, given that the phrase "land sparing" originally only applied to changes in land use between farmed and intact, wild habitats, and not within different areas of farms (Green et al., 2005).

Of course, the findings of this study are sensitive to how sustainable intensification has been defined and characterised. Our approach has focused on trends, rather than on whether or not the farms are, or have been, managed in a way that is sustainable in some objectively-defined manner (e.g. Gomez-Limon and Sanchez-Fernandez (2010)). This approach is consistent with the definition of sustainable intensification which focuses on increasing the supply of food and other ecosystem services whilst addressing neither capacity nor demand. The timescale was selected to detect changes in ecosystem services due to changes in farm systems and management, as opposed to annual variation. Such detection would be improved by using time-series of annual data from farms; a longer time series would also allow the detection of slower processes, such as changes to soils and levels of energy use following the introduction of zero tillage, and also changes in the resilience of ecosystem service provision.

The indicators used in this study relate to ecosystem services characterised as flows per unit area, using only data readily available to the farmer. Environmental stocks (natural capital) are not included; for example, the estimates of GHG emissions are not related to the potential value of the soil carbon stocks already present on the farm, while other indicators, such as soil erosion and landscape quality, were not considered to be measurable with adequate precision without additional field data. This analysis is sensitive to the choice and weighting of indicators (Bockstaller et al., 2009). This study has relied on models based on national data to estimate levels of pollution; such models require farm-scale validation in order to address farm-to-farm variation more precisely, and to take into account local innovations, such as the adoption of zero tillage. A time series of data is also necessary to be able to detect statistically significant change over and above the usual annual variation in food production and other ecosystem services.

Conclusions

This study demonstrates that some British farms have increased levels of food production and enhanced the quality of their environment over recent years; they have achieved sustainable intensification, as defined by this project. The drivers have been largely financial; farmers have sought to reduce input costs, thereby reducing wastes and pollution, while income streams from agri-environment schemes provided the incentive to enhance the farm for biodiversity. These drivers have played out differently between farm types. Such results are helpful when considering how policies can evolve to support sustainable intensification in appropriate situations.

This study also demonstrates that a simple approach to assessing sustainable intensification, using a small number of variables derived from data that farms already hold, is capable of distinguishing between different farming strategies, and so could form the basis of a much broader monitoring programme of agricultural sustainability. However, this approach is at the expense of excluding some possible indicators (e.g. landscape quality, species richness), and lack of sensitivity to farmer innovation.

There are many possible approaches to indicating sustainable agriculture (e.g. Rodrigues et al. (2010); Fumagalli et al. (2012); van Oudenhoven et al. (2012)); ultimately we need indicator suites that are validated using field data, transparent, inter-operable, capable of development, and that give accurate feedback to farmers, the food chain, policy makers and consumers that the food is being produced efficiently and sustainably.

Acknowledgements

This study was funded by the Land Use Policy Group, comprising Countryside Council for Wales (CCW), Natural England (NE), Scottish Natural Heritage (SNH), the Environment Agency, the Scottish Environmental Protection Agency and the Northern Ireland Environment Agency. We thank Brian Pawson (CCW), Graeme Kerr (NE), James Petts (NE), Cecile Smith (SNH), Maria de la Torre (SNH), Margaret Griffiths and Bryn Thomas (both Welsh Government) for their support and discussions of this work. We also wish to thank Jules Pretty, Ian Crute and Alan Buckwell for their comments on this work.

References

Baker, D.J., Freeman, S.N., Grice, P.V., Siriwardena, G.M., 2012. Landscape-scale responses of birds to agri-environment management: a test of the English Environmental Stewardship scheme. J. Appl. Ecol. 49, 871-882.

Bateman, I.J., Mace, G.M., Fezzi, C., Atkinson, G., Turner, K., 2011. Economic Analysis for Ecosystem Service Assessments. Environ. Resour. Econ. 48, 177-218.

Bockstaller, C., Guichard, L., Keichinger, O., Girardin, P., Galan, M.B., Gaillard, G., 2009. Comparison of methods to assess the sustainability of agricultural systems. A review. Agron. Sustain. Dev. 29, 223-235.

Butler, S.J., Brooks, D., Feber, R.E., Storkey, J., Vickery, J.A., Norris, K., 2009. A cross-taxonomic index for quantifying the health of farmland biodiversity. J. Appl. Ecol. 46, 1154-1162.

Butler, S.J., Vickery, J.A., Norris, K., 2007. Farmland biodiversity and the footprint of agriculture. Science, pp. 381-384.

Cardenas, L.M., Cuttle, S.P., Crabtree, B., Hopkins, A., Shepherd, A., Scholefield, D., del Prado, A., 2011. Cost effectiveness of nitrate leaching mitigation measures for grassland livestock systems at locations in England and Wales. Sci. Total Environ. 409, 1104-1115.

Cary, J.W., Wilkinson, R.L., 1997. Perceived profitability and farmers' conservation behaviour. J. Agric. Econ. 48, 13-21.

Chan, W., Brown, J., Lee, S.M., Buss, D.H., 1995. Meat, Poultry and Game. Supplement to McCance and Widdowson's 'The Composition of Foods'. The Royal Society of Chemistry and the Ministry of Agriculture, Fisheries and Food, London.

CLA, 2012. CLA CALM Calculator. CLA & Savilles.

DECC, 2012. UK Greenhouse Gas Emissions. Department of Enegy and Climate Change, London.

Defra, 2011. UK biodiversity indicators in your pocket 2011: Measuring progress towards halting biodiversity loss. Defra, London.

Defra, DARD (NI), Scottish Government RERAD, Welsh Assembly Government DRAH, 2012. Agriculture in the UK 2011. p. 124.

Del Prado, A., Misselbrook, T., Chadwick, D., Hopkins, A., Dewhurst, R.J., Davison, P., Butler, A., Schroder, J., Scholefield, D., 2011. SIMS(DAIRY): A modelling framework to identify sustainable dairy farms in the UK. Framework description and test for organic systems and N fertiliser optimisation. Sci. Total Environ. 409, 3993-4009.

Doxa, A., Paracchini, M.L., Pointereau, P., Devictor, V., Jiguet, F., 2012. Preventing biotic homogenization of farmland bird communities: The role of High Nature Value farmland. Agr. Ecosyst. Environ. 148, 83-88.

FAO, 2012. Latest forecasts confirm a significant tightening of global grain supply-demand balance in the 2012/13 marketing season: FAO Cereal Supply and Demand Brief September 2012. FAO, Rome.

Firbank, L., 2009. COMMENTARY: It's not enough to develop agriculture that minimizes environmental impact. Int. J. Agric. Sustain. 7, 151-152.

Firbank, L., Critchley, C.N.R., Fowbert, J.W., Fuller, R.J., Gladders, P., Green, D.B., Henderson, I., Hill, M.O., 2003. Agronomic and ecological costs and benefits of set-aside in England. Agriculture Ecosystems & Environment 95, 73-85.

Firbank, L., Bradbury, R., McCraken, E., Stoate, C., 2011. Enclosed Farmland. In: United Kingdom National Ecosystem Assessment Technical Report. UNEP -WCMC, Cambridge, pp. 197-240.

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

Francis, J.A., Vavrus, S.J., 2012. Evidence linking Arctic amplification to extreme weather in mid-latitudes. Geophys. Res. Lett. 39.

Fumagalli, M., Acutis, M., Mazzetto, F., Vidotto, F., Sali, G., Bechini, L., 2012. A methodology for designing and evaluating alternative cropping systems: Application on dairy and arable farms. Ecol. Indic. 23, 189-201.

Gomez-Limon, J.A., Sanchez-Fernandez, G., 2010. Empirical evaluation of agricultural sustainability using composite indicators. Ecol.Econ. 69, 1062-1075.

Gooday, R., Anthony, S., 2010. Mitigation Method-Centric Framework for Evaluating Cost-Effectiveness. Defra Project WQ0106 (Module 3). ADAS, Wolverhampton, p. 75.

Green, R.E., Cornell, S.J., Scharlemann, J.P.W., Balmford, A., 2005. Farming and the fate of wild nature. Science 307, 550-555.

Hanley, N., Acs, S., Dallimer, M., Gaston, K.J., Graves, A., Morris, J., Armsworth, P.R., 2012. Farm-scale ecological and economic impacts of agricultural change in the uplands. Land Use Policy 29, 587-597.

Hutchins, M., Fezzi, C., Bateman, I., Posen, P., Deflandre-Vlandas, A., 2009. Cost-Effective Mitigation of Diffuse Pollution: Setting Criteria for River Basin Management at Multiple Locations. Environ. Manage. 44, 256-267.

Krebs, J.R., Wilson, J.D., Bradbury, R.B., Siriwardena, G.M., 1999. The second silent spring? Nature 400, 611-612.

McIntyre, B.D., Herren, H.R., Wakhungu, J., Watson, R.T. (Eds.), 2009. Agriculture at a crossroads: International Assessment of Agricultural Science and Technology for Development Global Report. Island Press, Washington.

Mouysset, L., Doyen, L., Jiguet, F., 2012. Different policy scenarios to promote various targets of biodiversity. Ecol. Indic. 14, 209-221.

Pretty, J., Smith, G., Goulding, K.W.T., Groves, S.J., Henderson, I., Hine, R.E., King, V., van Oostrum, J., Pendlington, D.J., Vis, J.K., Walter, C., 2008a. Multi-year assessment of Unilever's progress towards agricultural sustainability I: indicators, methodology and pilot farm results. Int. J. Agric. Sustain. 6, 37-62.

Pretty, J., Smith, G., Goulding, K.W.T., Groves, S.J., Henderson, I., Hine, R.E., King, V., van Oostrum, J., Pendlington, D.J., Vis, J.K., Walter, C., 2008b. Multi-year assessment of Unilever's progress towards agricultural sustainability II: outcomes for peas (UK), spinach (Germany, Italy), tomatoes (Australia, Brazil, Greece, USA), tea (Kenya, Tanzania, India) and oil palm (Ghana). Int. J. Agric. Sustain. 6, 63-88.

Pretty, J., Toulmin, C., Williams, S., 2011. Sustainable intensification in African agriculture. Int. J. Agric. Sustain. 9, 5-24.

Rodrigues, G.S., Rodrigues, I.A., Buschinelli, C.C.D., de Barros, I., 2010. Integrated farm sustainability assessment for the environmental management of rural activities. Environ. Impact Assess. Rev. 30, 229-239.

Rounsevell, M.D.A., Annetts, J.E., Audsley, E., Mayr, T., Reginster, I., 2003. Modelling the spatial distribution of agricultural land use at the regional scale. Agr. Ecosyst. Environ. 95, 465-479.

Royal Society, 2009. Reaping the benefits: Science and the sustainable intensification of global agriculture. Royal Society, London, p. 72.

UKNEA, 2011. United Kingdom National Ecosystem Assessment: Synthesis of the Key Findings. UNEP-WCMC, Cambridge.

van Oudenhoven, A.P.E., Petz, K., Alkemade, R., Hein, L., de Groot, R.S., 2012. Framework for systematic indicator selection to assess effects of land management on ecosystem services. Ecol. Indic. 21, 110-122.

Table 1

Percentage composition of generic compound formulations for animal feeds. These values were used to estimate the amount of crops required to make up the diets of animal feeds imported to the farm, which were then used to estimate the land required to grow the animal feeds, using average UK yield data. Minor feed constituents are not included.

	Dairy	Beef	Sheep	Fattening pigs	Poultry
Wheat	15		10	45	45
Barley	20	40	22	20	20
Wheat and other cereals	20	21	15	9	9
Oilseed rape meal	20	20	10	11	11
Soybean cake and meal	5		5	7	7
Sunflower cake and meal			5		
Field beans	5		10		
Sugar beet feed, dried molasses	8	12	15	3	3

	Points for stock	Points for change
Designated sites		
Connectivity of species-rich grassland increased / no change / decreased		-1/0/+2
Native livestock breeds		
Presence of native breeds	3	3
Numbers of native breeds increased / no change / reduced		+1/0/-1

Table 2

Energy content of agricultural produce, used to estimate energy content of food produced from the farms. FW = fresh weight; DM = dry matter. Data are compiled from several sources, i.e. (Chan et al., 1995); The Dairy Council (undated); US FDA (undated) from the UK Nutrient Databank, maintained by the Food Standards Agency (FSA) (REF). All meat weights relate to raw, trimmed lean portions, supplemented by estimates of the energy content of the non-consumable proportions of livestock (Kempster et al 1985; BPEX 2012; EBLEX 2012).

Energy content

Beef 5.7 GJ/t FW Lamb 6.5 GJ/t FW Pork 5.2 GJ/t FW Poultry 5.2 GJ/t FW Milk 2.8 GJ/1,000 l Wheat 18.4 GJ/t DM Field beans 18.6 GJ/t DM Sugar beet 14.0 GJ/t DM Potatoes 13.0 GJ/t DM Vegetables 6.0 GJ/t DM Soft fruit 7.1 GJ/t DM

	Dairy	Beef	Sheep	Fattening pigs	Poultry
Wheat	15		10	45	45
Barley	20	40	22	20	20
Wheat and other cereals	20	21	15	9	9
Oilseed rape meal	20	20	10	11	11

Table 3

The scoring system for biodiversity. Few farmers have direct information on biodiversity stock and change, so the assessment used here follows the UK Biodiversity Indicators by giving points for habitats, features, management practices and overall farm management. Scores are generated for stock at baseline, and change between baseline and latest data. For biodiversity stock, the points were summed, and then set to a five point scale as follows: 0 points was set to 1 (every farm has at least some biodiversity); 1-5 was scaled as 2; 6-10 was scaled as 3: 11-15 was scaled as 4; 16-20 was scaled as 5. Change can be positive or negative, and so was rescaled as follows: -20 to -11 was scaled as -2: -10 to -3 was scaled as -1: -2 to +2 was scaled as 0: 3-10 was scaled as +1; 11-20 was scaled as +2. See text for details.

Dairy	Beef	Sheep	Fattening pigs	Poultry

Sites of Special Scientific Interest and equivalent 1 per site 1 per site

Priority Habitats

Number of priority habitats present 1 per 1 per habitat habitat

Agri-environment schemes

Participation in Entry level scheme 1 1 Participation in Higher Level scheme 3 3 For each scheme option 1 1

Member of other schemes or initiatives

Farm assurance 1 1 Farm environment 1 1 Biodiversity records 1 1

Land management

More spring sown / no change / more autumn sown crops +3 / 0 / -3

More hay / no change / more silage for animal feed +3 / 0 / -3 Area of non-cropped habitats increased / no change / +3 / 0 / -3 reduced Area of drained land reduced / no change / increased +3 / 0 / -3

Livestock density reduced / no change / increased +3 / 0 / -3 Crop diversity increased / no change / reduced +3 / 0 / -3 Increased / no change / reduced diversity of non-cropped +3 / 0 / -3 habitats Set-aside arable land created / no change / reverted to +3 / 0 / -3 crops

Table 4

Changes in SI indicator scores for each case study farm in 2011, expressed as percentages of the scores in 2006. Bold type and white background is an enhancement (an increase in food production and biodiversity score, a reduction in emissions); normal type and pale grey background indicates that change is within 10 % for numerical data, and 0 for biodiversity; these are interpreted as little or no significant change; italic type and darker grey background represents a deterioration.

A **33**

Dairy Beef Sheep Fattening

ra *-19*

bl

e **18**

pigs

Wheat 15 10 45

-12

Barley 20 40 22 20

15

-14

-1 *19* **-1**

10

4

-11

Mixed **52**

20

14 -1

-1

26

3

Dairy 5

11

11

-2

10 **3**

5 *15*

Livestock **35** *28*

A2 A3 A4 -4 -5 4 5

A5 A6 A7 **-27** -

M1 M2 3

M3 M4

5 *17* *49* *33* **1**

D1 D2 D3 -2

35 *13* *15* *39* 0

D4 L1 L2 -5

L3 L4 -8

-

11

2 *11* 1 8 -

5

-
1
8

-13

4 1 9 4

21

-18

-2
-30

47

-10 -9

-12

-8 0 8

-27

9

1

30

-20

-4

1 1 2 2

-*1*

0 0 0

36

1 1

29

-*1*

0 0 0

2 1

0 0

L5

-15 -23
-43 2

Figure Legends

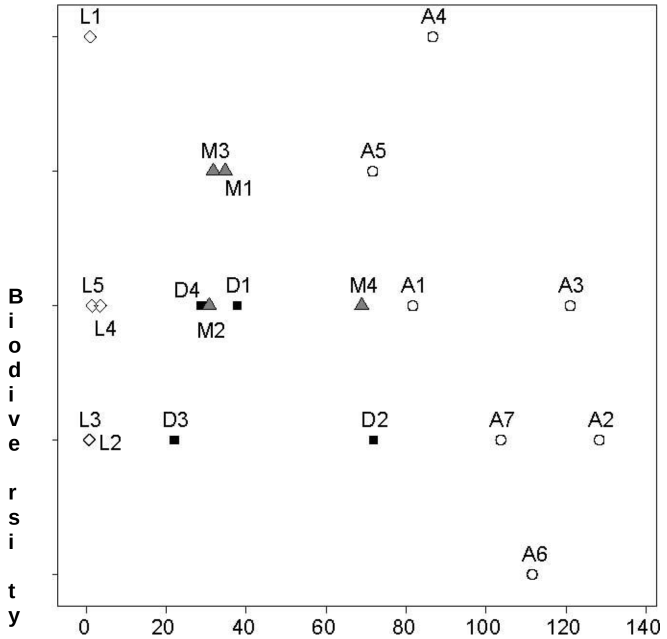
Fig. 1: Relationships between emissions of ammonia and GHGs, losses of nitrate and biodiversity scores and food production for 20 case study farms at the baseline observation date (usually 2006). The farms are labeled individually, and coded according to farm type: arable farms are prefixed “A” and shown as open circles; dairy prefixed “D” and shown as black squares; mixed farms are prefixed “M” and shown as grey triangles and upland livestock farms in Less Favoured Areas are prefixed “L” and shown as open diamonds. Ammonia and nitrate emissions are in Kg ha^{-1}; GHG emissions are kg CO_2equiv/ha; food is GJ^{-1}. GHG emissions data are missing for farm A5.

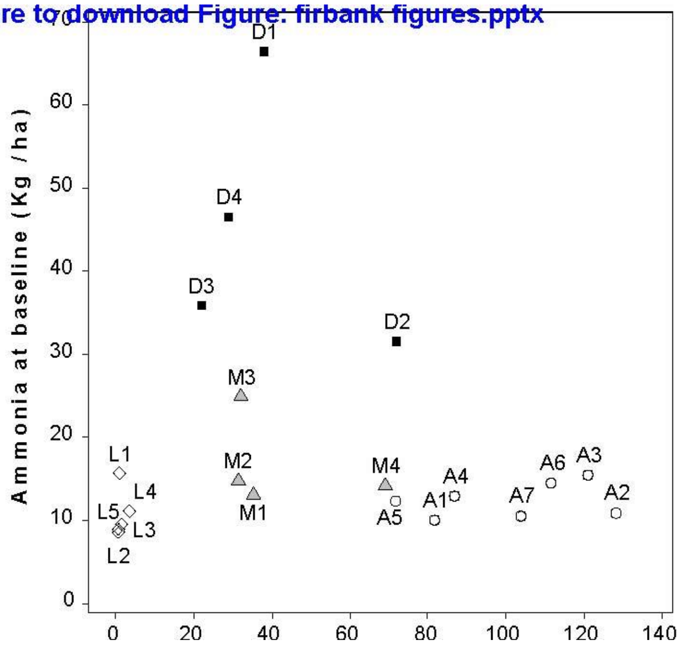
Fig. 2: Percent changes in food production between baseline (usually 2006) and latest data (usually 2011) compared with food production at baseline, expressed as GJ ha^{-1}. Farms are identified as Figure 1.

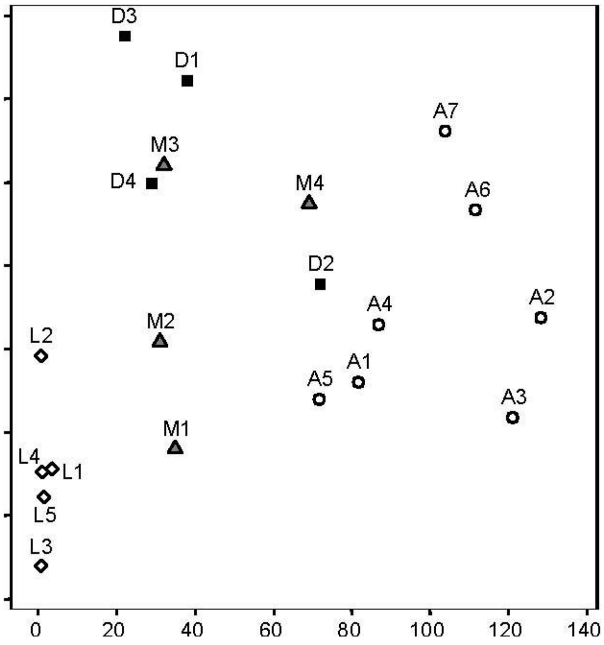
Fig. 3: Concomitant changes in food production and environmental variables on the case study farms between baseline and latest data. All changes are expressed as percentages of values at baseline, and visualized so that environmental enhancements are above the horizontal line (increases in biodiversity scores, reductions in emissions of ammonia, GHGs and losses of nitrate) and increases in food production are to the right of the vertical line, i.e. win-win situations are always shown in the top right quadrants. Farms are identified as in Fig 1. GHG emissions data are missing for farm A5.

Fig. 4: Changes in emissions of ammonia, GHG and losses of nitrate, between baseline and latest data, expressed per unit energy content of food (KJ ha^{-1}). The sloped lines represent no change in emissions per unit food energy. Log scales are used to cope with the very wide of emissions over the farm types. Farms are identified as in Fig 1. GHG emissions data are missing for farm A5.

Figures


5
4
3
Biodiversity
score at baseline
L1
A4
M3
A5
M1
L5
D4
D1
M4
A1
A3
L4
M2
L3
D3
D2
A7
A2
L2
A6
0
20
40
60
80
100
120
140


70
D1
60
50
D4
40
D3
D2
30
M3
20
L1
M2
M4
A4
A6
A3
L4
A1
A7
A2
10
L5
L3
M1
A5
L2
0
0
20
40
60
80
100
120
140
Ammonia at baseline (Kg /ha)


D3
D1
A7
M3
D4
M4
A6
D2
A4
A2
L2
M2
A1
A5
A3
M1
L4
L1
L5
L3
0
20
40
60
80
100
120
140
2
1
0
Nitrate losses (Kg/ha)

8 0

GHG emissions at baseline (K

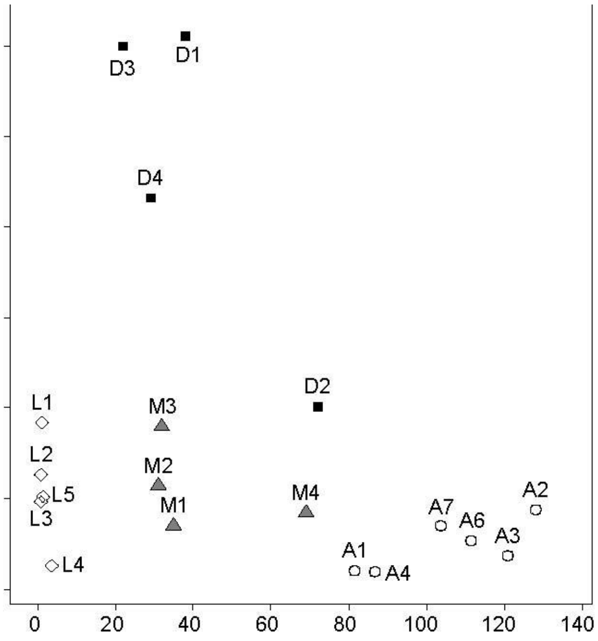
0

6 0

5 0

4 0 1 0 0 0 0

8



g CO₂ e/ha)

1 4 0 0 0

0 0 0

6 0 0 0 3 0

2 0 4 0 0 0

7 0

1 2 0 0

2000

Food production at baseline (KJ / ha) Food production at baseline (KJ / ha)

**C
H
a
n
g
e
in
fo
o
d
p
r
o
d
u
ct
io
n
(
%
)**

6
0

5
0

4
0

3

0

20

10

0

-

1

0

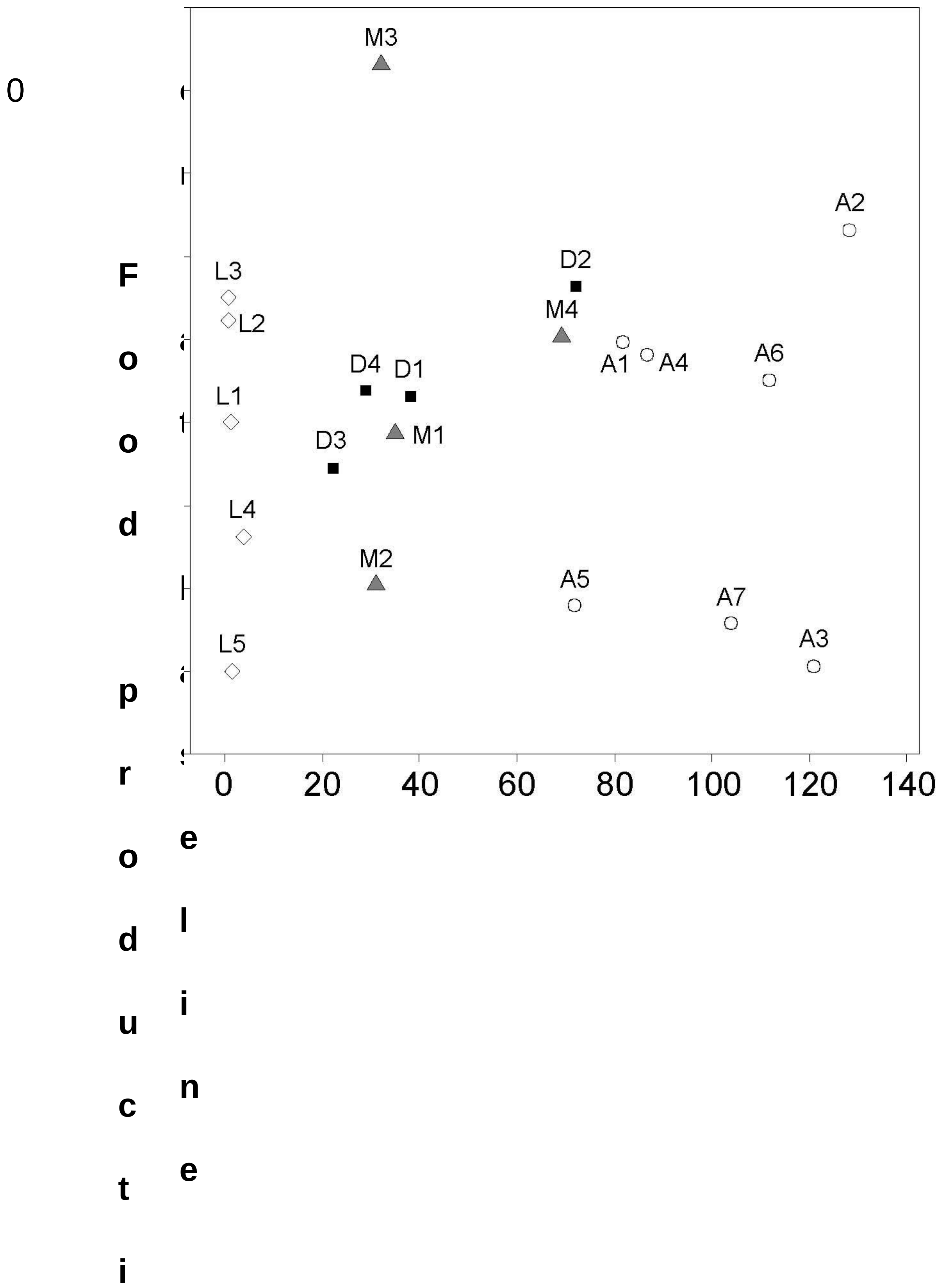
-

2

0

-

3


0
F o o d p r o d u c t i
e l i n e
M3
A2
D2
L3
L2
M4
A1
A4
A6
D4
D1
L1
M1
D3
L4
M2
A5
A7
L5
A3
0
20
40
60
80
100
120
140

(

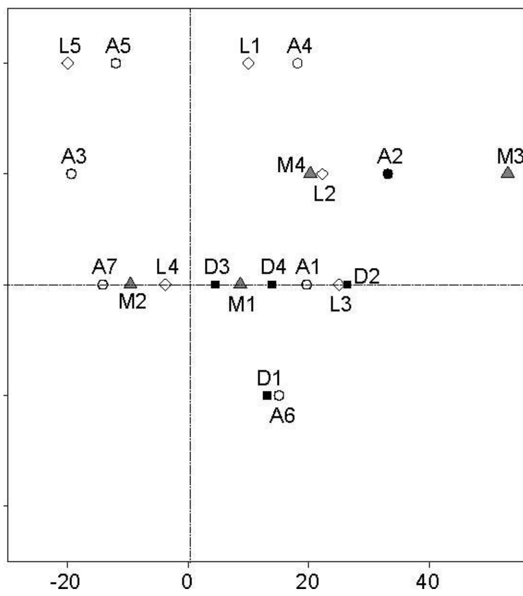
K h

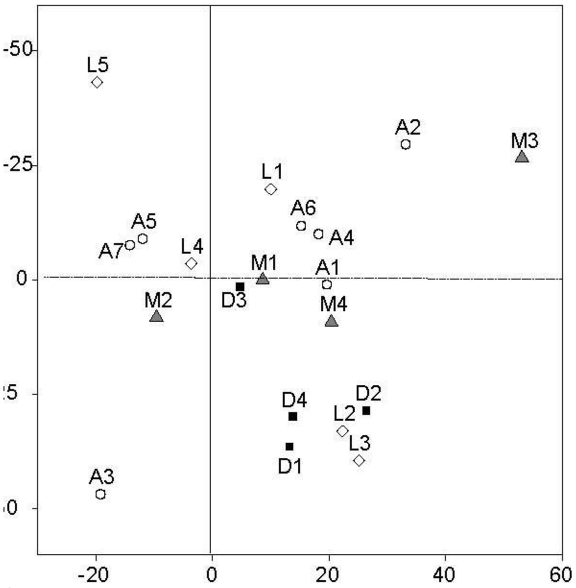
J a

)

/

-50





2

1

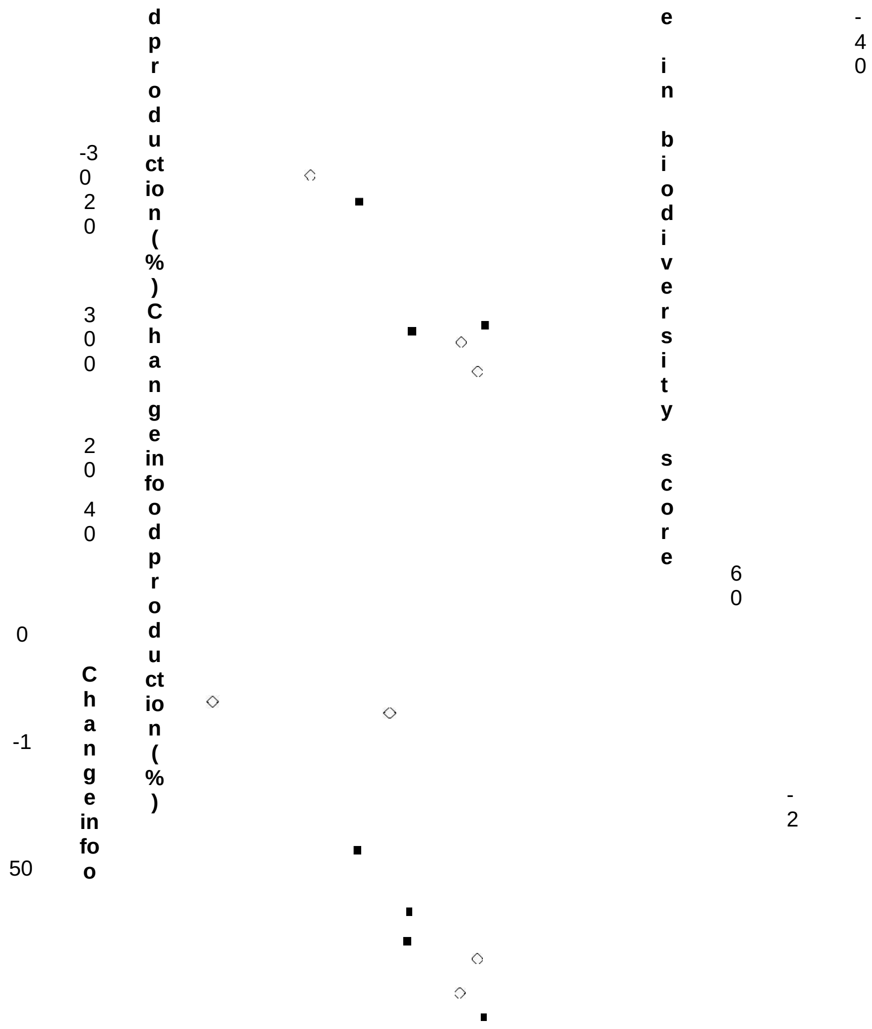
Change in GHG emissions (%)

Change in nitrate losses (%)

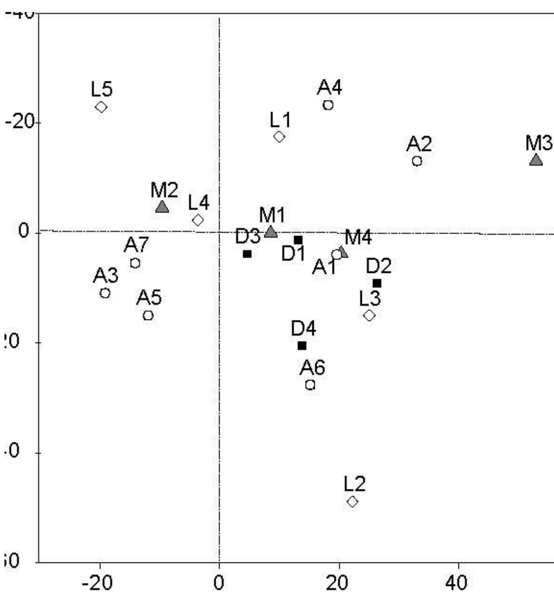
n ammonia emissions (%)

Change i

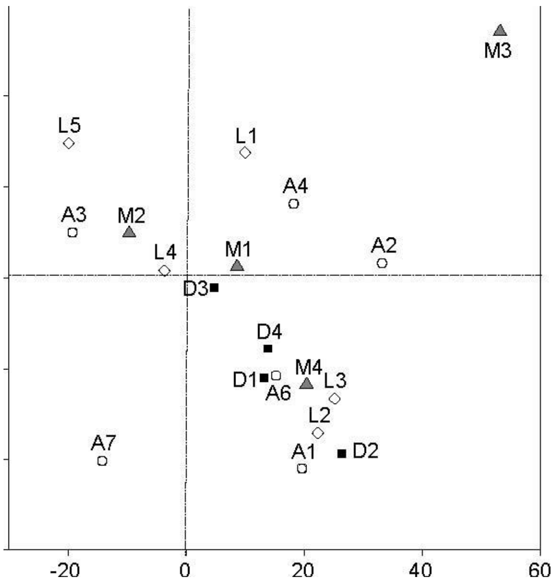
Chang



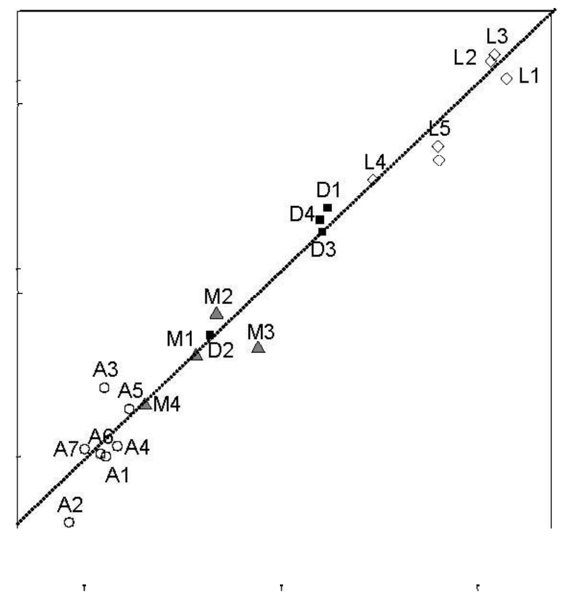
-30
20
300
20
40
0
-1
50
Change in food production (%)
Change in food production (%)
Change in fo o
e in biodiversity score
-40
60
-2

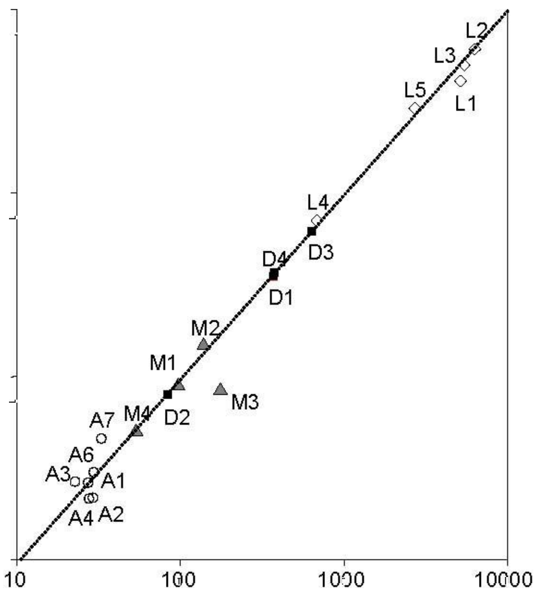


-20
0
-20
0
20
40
L5
A4
L1
A2
M3
M2
L4
M1
D3
D1
M4
A7
A1
D2
A3
A5
L3
D4
A6
L2



M3
L5
L1
A4
A3
M2
L4
M1
A2
D3
D4
M4
D1
A6
L3
L2
A7
A1
D2
-20
0
20
40
60

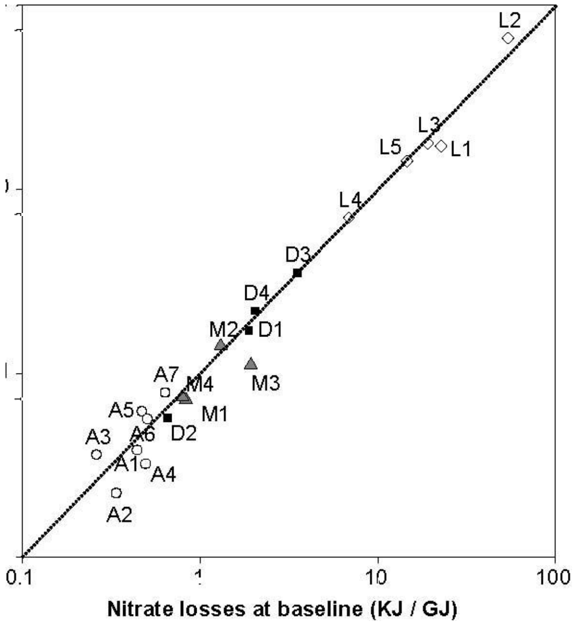




10000

L2

1
0



0.1 1 10

GHG emissions at baseline (Kg CO$_2$e / GJ)

L
1

A
1
M
4
1
0
0
M
3

Ammonia emissions at baseline (Kg /GJ)

10

Latest n nitrate losses (Kg / GJ) Latest ammonia emissions (Kg / GJ)
1000

L4

1

Latest GHG emissions (Kg CO$_2$e/ GJ)
D1 D1

A
7

A
3
A
2

A3

A2