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Is it possible to increase the sustainability of arable and ruminant agriculture by reducing inputs?

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To Agricultural Systems

Abstract

Until recently, agricultural production was optimised almost exclusively for profit but now farming is under pressure to meet environmental targets. A method is presented and applied for optimising the sustainability of agricultural production systems in terms of both economics and the environment. Components of the agricultural production chain are analysed using Environmental Life-Cycle Assessment (LCA) and a financial value attributed to the resources consumed and burden imposed on the environment by agriculture, as well as to the products. The sum of the outputs is weighed against the inputs and the system considered sustainable if the value of the outputs exceeds those of the inputs. If this ratio is plotted against the sum of inputs for all levels of input, a diminishing returns curve should result and the optimum level of sustainability is located at the maximum of the curve. Data were taken from standard economic almanacs and from published LCA reports on the extent of consumption and environmental burdens resulting from farming in the UK. Land use is valued using the concept of ecosystem services. Our analysis suggests that agricultural systems are sustainable at rates of production close to current levels practiced in the UK. Extensification of farming, which is thought to favour non-food ecosystem services, requires more land to produce the same amount of food. The loss of ecosystem services hitherto provided by natural land brought into production is greater than that which can be provided by land now under extensive farming. This loss of ecosystem service is large in comparison to the benefit of a reduction in emission of nutrients and pesticides. However, food production is essential, so the coupling of subsidies that represent a relatively large component of the economic output in EU farming, with measures to reduce pollution are well-aimed. Measures to ensure that as little extra land is brought into production as possible or that marginal land is allowed to revert to nature would seem to be equally well-aimed, even if this required more intensive use of productive areas. We conclude that current arable farming in the EU is sustainable with either realistic prices for products or some degree of subsidy, and that productivity per unit area of land and greenhouse gas

emission (subsuming primary energy consumption) are the most important pressures on the sustainability of farming.

Keywords

Sustainable Agriculture, Total factor productivity, Environment, Environmental burden, Resource use, Environmental economics, arable, ruminant, Life Cycle Assessment, Ecosystem services.

1. Introduction

Formerly, agricultural production was optimised almost exclusively for farm profit. Latterly, however, farming has come under increasing pressure to meet environmental targets (Goulding et al., 2008). An imbalance between fertiliser supply and crop offtake as well as soil erosion may lead to the loss of nutrients to air and water; sorbed pesticides may wash into natural waters, and energy consumption at all stages of agricultural production contributes to global warming. If agricultural production is to be truly sustainable, it makes sense to weigh economic benefits against environmental burdens and the consumption of resources. It is difficult to do this on a consistent basis without attributing a cash value to the environmental impacts, however. Imperfect though this is, we present methodology to make such a comparison in a transparent and objective way.

Given knowledge about the extent of farming in the UK, it is possible to approximate the contribution of each farming system to the total environmental burden. Pretty et al. (2005a&b, 2003 and 2000) attributed environmental costs to the various components of agriculture for the UK as a whole, Hartridge and Pearce (2001) reviewed the environmental effects of farming in the UK in economic terms, and Atkinson et al. (2004) examined the potential of monetised accounting of the environmental effects of agriculture.

Environmental Life Cycle Assessment, LCA, (http://www.iso-14001.org.uk/index.htm) seeks to take account of all the inputs to and outputs from a production system in order to take a complete view within defined system boundaries.

The primary inputs are traced far back along the production system: e.g. small components of oil extraction and refining or iron ore mining and steel production are attributed to the annual use of a tractor in agricultural production. Costs in this sense are taken to be environmental costs or burdens as well as financial costs. LCA normally assembles these separately into their own categories. Using such an approach, Williams et al. (2006) have published a thorough LCA of several commodities produced within UK agriculture. Here we convert all LCA components into monetary units in order to express them on a single, economic basis.

Total Factor Productivity (TFP) is the ratio of the economic outputs from a system to the inputs (Lynam and Herdt, 1989; Ehui and Spencer, 1992; Barnett, 1994). Barnett et al. (1995) showed how this concept could be used to include environmental considerations by attributing a cost to each of the resources and to the effects of each burden on the environment. TFP is used as an index and normally calculated at the optimum yield response.

High-input farming is geared to achieving maximum profit. This often implies levels of production just short of the physiological optimum response of the plant or animal to inputs. Beyond this point, increasing inputs and therefore costs achieve small increases in yield only which are insufficient to pay for the extra inputs. This suggests, however, that in the region of this optimum substantial reductions in input might be achieved with little loss of yield or profit. Also, if one input, e.g. nitrogen is reduced then less of other inputs may be needed. Despite much work on reduced-input farming, little has been done to establish the optimum *level* of reduction. Implicit in this idea, however, is the assumption that the rate of consumption of environmental services and the rate of pollution reduce along with a decrease in the rate of intensity.

Our objective in this article is to develop and use methodology for estimating the optimum level of all inputs in any given system of production that reduces as much environmental pollution as possible for least consumption of resources within the constraint of maintaining farm income at as high a level as possible. We do this by plotting TFP against the total inputs, including environmental inputs, and deduce the optimum in the likely sustainability of each of several agricultural systems to be at the maximum of the curve. In doing so, we try to include estimates for the cost or value of all components in a transparent way. Recent fluctuations in the costs of inputs and farm commodities persuaded us that the idea of a trend with time was meaningless unless the variability is itself indexed (Lien et al., 2007). Accordingly we explore the underlying structure of sustainability in what is essentially a static measure of the components of farming that are likely to determine sustainability over time. All data and calculations are included in spreadsheets that are available at www.rothamsted.bbsrc.ac.uk/aen/TFP/. If better values become available and are agreed upon by the scientific community, the spreadsheets can be updated accordingly. In addition, we analyse the make-up of the environmental costs and show how these change with changing intensity of farming.

2. Methods

2.1 Calculation system

A way of examining the sum economic value of an activity by expressing all components on the same basis, is to analyse the Total Factor Productivity (TFP; Barnett et al., 1994). This is the value-weighted sum of the outputs from a farming system divided by the cost-weighted sum of the inputs.

$$TFP = \frac{\sum_{j=1}^{m} P_j Q_j}{\sum_{i=1}^{n} W_i X_i} \qquad [1]$$

where W_i is the cost of each of n input factors used at rate X_i, and P_j is the value of each of m outputs yielding a quantity Q_j each. If TFP is greater than 1.0 and remains so for a number of years a system can be said to be sustainable economically. The index can be used to assess the decline in viability or the progressive benefits of adopting more sustainable practices, but presupposes that the intention is to continue farming and maintain the production of food, as we explain below. A purely economic analysis without factoring in the environmental costs would be biased (Barnett et al., 1994). Therefore environmental costs, such as greenhouse gas (GHG) emissions and nitrate leaching are factored in as additional input costs (Barnett et al., 1995). The alternative of including them as output penalties might lead to a negative value for the index. It should be noted that we classify farm support (i.e. subsidies) as an output because it contributes to field income and therefore contributes to profitability. Support, either of production or of an environmentally beneficial measure, is easily included as its financial incentive, P, in relation to unit environmental target, Q. This provides a logical and straightforward way of investigating the response of all outputs to all inputs, and enables us to assess the importance of such support to the sustainability of any system.

Responses change with inputs and it is our thesis that a maximum in the TFP versus inputs graph can be found, i.e. that there is an optimal system. Since this value of the TFP index and the input costs include the environmental burdens, the maximum should represent the optimum level of intensity of production that balances environment with productivity. Note that the analysis proposed may not explain farming strategy since it is usually net profit (i.e. the difference between the numerator and denominator in Eq [1] multiplied by the volume but without the environmental factors) that determines what a farmer does.

LCA is defined for a system. Our system includes stages prior to the farm but excludes everything once the product is sold and leaves the farm; in other words transport, processing, packaging and distribution. Direct costs for the production of agricultural chemicals are not included in our analysis because they are included in the price paid by the farmer and appear in the denominator of the TFP index. We therefore depart from the norm set for LCA . We do, however, apportion the environmental costs of the GHGs emitted in the production of agricultural chemicals and other environmental costs.

2.2 Environmental costs

Economists refer to costs that do not appear in their calculations as 'external'. Examples are the environmental burdens and uncosted consumption of resources. Because we wish to internalise these costs we refer to them as environmental costs and have avoided the term external. Besides an analysis of the TFP response to inputs, we provide a breakdown of the individual environmental costs at different rates of input. Indirect environmental costs, associated with chemical and machinery production or the construction of buildings, are less easy to attribute and here we have relied on the LCA analysis of Williams et al. (2006). A full description of the data we have used and the ways in which we have processed them is too detailed to include in the main body of this article. Full details are provided in the supplementary information included on the web with this article and in Williams et al. (2006). Only the essential elements are given below.

2.2.1 Primary energy

The prices of energy can be stated accurately. Direct energy costs for farm operations were set at those current at the end of January 2006 as detailed in the supplementary information. These include fuel for machinery and electricity used in drying or cooling harvested produce. Energy costs have risen sharply since that date, however.

The cost of embodied energy in indirect inputs is accounted for in their cash cost. The energy used for manufacturing fertilisers, pesticides or machinery for arable costs is thus indirectly implied by their cost. The consumption of primary energy is thus limited to what we have called operational costs such as fuel to power tractors or the drying of harvested grain. On the other hand, environmental emissions associated with manufacture were given environmental costs using the emission values per unit input from Williams et al. 2006 and the costs of Pretty. The same argument applies to feeds imported to livestock farms.

2.2.2. Pesticides, herbicides and other chemical control agents

Besides the economic cost and environmental burden of producing these chemicals (primary energy, etc.), their use is itself an environmental burden. We have estimated this burden as the sum of the costs of removing the compounds from drinking water, costs to farmers and the National Health Service of acute damage to human health, and the cost of the loss of abundance and diversity of wildlife. The costs of pesticides to human health are thought to have been considerably underestimated as they do not include chronic effects (e.g. cancers) and acute effects may well be under reported (Pretty et al. 2000). In contrast, however, Trewavas (2004) avers that exposure to manufactured pesticides and sprays is associated with lower rates of cancer than in the general population. Notwithstanding this debate, Pretty et al. (2005a and 2000) estimate environmental costs of chemicals for the whole of the UK. We use their data, expressing them per hectare or per kg commodity by attributing the UK pesticide costs first of all to commodities based on their relative production rates and on the make up of a typical range of sprays used with each commodity, as explained in detail in the appendix. Based on experimental results, cereal yields given by the Wheat Disease Manager (Audsley et al. 2005) improve if sufficient amounts of bioicidal chemicals of the correct kind are applied. We have chosen to invert this relationship and so have derived the response at reduced applications. The national burden can then be partitioned to crops at different rates of input. Chemicals are assumed to be applied even if fertilisers are not. Reductions in chemical inputs are obtained by reducing the number of sprays and accepting some actual reduction or risk of reduction in crop yield from weeds, pests and diseases. Reducing the concentration of active ingredient in a spray is not recommended because of the danger that the target will develop resistance. We have not, therefore, used reduced concentrations in our calculations.

2.2.3. Eutrophication

The financial burden associated with nitrogen and phosphorus loss from agriculture has been expressed on a national basis by Pretty et al. (2005a&b 2003 and 2000,). This cost is partly the removal of the nutrients from drinking water but also of eutrophication, loss of biodiversity and habitat, and costs associated with the unsightly appearance of algal blooms that diminish the value of water-side properties, of amenity and recreation, and thus also the tourist trade. These data were attributed to farming as a whole and related to current, average fertiliser and crop use on farms, although we accept that a change in the use of P and to some extent N will be buffered in soil and will not immediately be reflected in emissions. The LCA norm assumes equilibrium conditions (i.e. projecting the outcomes of long-term farm practices) so our results must be seen as reflecting steady-state rather than the more dynamic results of an alteration to land-use or farming practice.

2.2.4. Global warming

The main GHGs carbon dioxide CO_2, methane CH_4 and nitrous oxide, N_2O are all emitted during agricultural production and to varying extents during the manufacture of inputs used in production. A large variation can be seen in the published values of GHG emissions and burdens (Table 1, Hartridge and Pearce 2001; Pretty et al. 2005a; Clarkson and Deyes, 2002; Atkinson et al. 2004). The latter two sets of authors argue that damage done by the longer-lived gases should not be referred to a global warming potential (GWP) of CO_2 equivalents, because the reference gas, CO_2 itself changes in concentration with time. To do so would inflate the value of a shorter-lasting gas such as methane. On the other hand the cost of damage today will be less than damage in future under the assumption that inflation consistently reduces the value of money, thus inflating the economic damage of longer-lasting gases in todays' terms. We use the estimates of the economic damage from GHG emission given by Atkinson et al. (2004). A small allowance is made for methane oxidation by soil. Strictly this should be given as an ecosystem service (section 2.2.5) but is already included in calculations within our source data (Williams et al, 2006).

2.2.5. Land-use

It is essential to take account of the area of land used in production because, although a less intensive system may pollute less on a per hectare basis, it requires more land area to produce the same amount of food. If extra land is needed to produce food with less pollution, where will that land come from and what will it cost? We have valued land using Costanza et al's (1997) ecosystem services approach. Cropland, grassland and temperate forest are given values for their environmental benefit, but we have discounted the value of their food and fibre production given by these authors because this residual benefit, for say cropland, is attributed to production in our analysis; that is to say it is included as an output in the numerator of the TFP index (Eq. 1). The cost is added to the denominator and is calculated from the value of the area of land lost from the substitute system: in all cases we assume forest is converted to agricultural land. To an extent, the value of land is included in an orthodox economic analysis because the land will cost a farm business rent or interest. These direct costs are included in our analysis. If more land is needed, we charge at the rate attributed to the ecosystem services provided by temperate forest (Costanza et al., 1997). We then proceed to analyse the system in two ways. Firstly, in estimating the cost of the consumption of land on a per hectare basis, we give the extra cost relative to the land-use at the optimum economic return, i.e. the marginal increase in land use. Thus land-use at optimum has a value of zero attributed to it on a per hectare basis. This is because we assume in our analysis that food-production at current rates is necessary and we refer our results to this norm. Secondly, however, in expressing the results on a per tonne of production basis, we give the actual ecosystem service cost attributed by Costanza et al (1997) to the land consumed in order to produce each tonne of that commodity.

2.3. Response to inputs

The well-known law of diminishing returns applies to crop production (e.g. Addiscott et al., 1991). Most usually this is seen with respect to nutrients and to nitrogen fertiliser in particular. We modelled crop yield using a response curve derived from the Quadmod system (ten Berge et al., 2000) because this links nitrogen uptake with response and application rate. The choice of a different response curve might make a small difference to the amounts of yield. We have re-parameterised Quadmod for the arable crops used in this analysis with data from our own experiments in the UK, as detailed in the supplementary information. Where our study has concentrated on farming close to the economic optimum, the calculations include benefits from economies of scale and we have used data pertaining to efficient production (e.g. ABC, 2005; Nix, 2005).

Extensification of ruminant systems was modelled by changing nitrogen fertiliser input to the grazing system and modifying the stocking rate to ensure a constant liveweight gain per head. The import of concentrates per unit grazed area was adjusted in proportion to the change in stocking rate. Thus, diets were not changed.

3. Results

We deal with the commodities in two groups: arable crops and finishing of ruminant meat.

3.1 Arable crops

3.1.1. Wheat

In Fig 1a we plot the wheat grain yield (tonnes ha^{-1}) and TFP index against total costs (variable, fixed and environmental). Our TFP index has a broad maximum at a cost of about £20-25 ha^{-1} less than that needed to obtain the physiological maximum. Note that this saving is largely in environmental benefits and not a reduction in farmer's costs. The reason for the lack of a sharp peak is to be found in the environmental costs (Fig 1b). Although these are small in relation to income and production costs, the increased need for extra land to maintain production with reduced inputs increases the sum of the environmental costs at the lower levels compared with optimum production. At its maximum, the TFP index is above one, if not greatly so and the system is broadly sustainable. However, support under the EU single farm payment scheme makes up a considerable proportion of the outputs (25% for wheat, for example), but applies to all levels of production. Recent increases in grain and oil prices would have a major impact on the results and the need for subsidies. Fig 1b suggests that, in operating at the optimum level for production, conventional wheat production is also operating close to the optimal use of environmental resources.

3.1.2. Oil seed rape (OSR)

The TFP index for OSR is barely 1 at its maximum (Fig 1c), although it should be noted that the TFP index excluding environmental costs was greater than unity near the maximum yield of the crop (data not shown. The maximum in the TFP occurs short of the physiological optimum as expected and represents a saving of about £40 ha^{-1}. The penalty from bringing extra land into production is irregular at low levels of OSR production (Fig 1d). If OSR is to be grown, the application of a small amount of fertiliser N increases saleable product greatly and so decreases the consumption of land relative to a crop receiving no N disproportionately (Fig 1c). The optimum production level is predicted to be close to the environmental optimum, but in this case somewhat less than current practice. There is, however, a demand for rape oil for biodiesel so this demand may have an increasingly positive effect on the TFP index.

3.1.3. Maincrop potatoes

The form of the potato response to inputs (Fig 1e) is similar to that of wheat. Production costs are high relative to environmental costs, however, and it is understandable why farmers do not judge it economic to reduce inputs even taking the cost of the environmental burdens into account. Note, however, the much larger total cost per hectare compared with the other two arable crops (Fig 1f). Apart from any other factors, root crops always require more energy per hectare than combinable crops, because deep ploughing is essential in cultivation and the soil must be worked again at harvest. With potatoes, the saving in moving back to the TFP maximum is several hundred pounds: mostly in environmental costs. A large environmental burden with this crop, however, is the GHG cost of storing tubers after harvest (Fig 1f).

3.2. Meat finishing systems

Animal production systems are much more complicated to analyse than the three arable systems in Figure 1. For example, a beef production system involves the initial production of calves, from either a dairy system or beef suckler system, each with its own burdens from inputs such as, feeding and housing. These are affected by fecundity, longevity, grassland management and feed conversion efficiency. The beef cattle are fed on a combination of feeds, generally including grass, silage and a range of concentrates (e.g. wheat, barley, wheatfeed oilseed meal and legumes). These all have their own inputs and burdens of production. There are also the associated outputs, such as manure, wool and leather. However, we did not include the value of the latter two products. Housing of the animals, either intensively or extensively, involves further inputs and burdens. There are many options for reducing inputs in such a system, e.g. using different combinations of feed stuffs in the concentrate mix, feeding over a longer period, so that the daily live weight gain is reduced and it takes longer for the animal to reach maturity, or reducing the ratio of concentrates to grass/silage. There are also opportunities for reducing inputs to the production of feedstuffs, principally nitrogen fertiliser, but which will then require a larger area of land to grow the concentrates or grass. We have not looked at all the above inputs simultaneously, but instead have decided to concentrate on N inputs to grassland (NCYCLE, Scholefield et al., 1991), in the production of grass grazed by ruminants, as an example of how inputs could be adjusted, and the resultant effects on environmental burdens. The range of N inputs encompasses those recommended in the UK (MAFF, 2000). For reference, however, the amount of N applied to grassland systems grazed or fed to beef is usually of the order of 100 kg N ha^{-1} with a maximum of about 250 kg N ha^{-1} in the UK (Defra, 2006). The meat production systems analysed here only deal with the finishing stage and do not include the breeding phase, which generally uses lower inputs.

3.2.1. Beef

We selected and have analysed the system known as 18-month beef, which relies on intensive grazing of fresh leys and good quality silage (see Nix, 2005, p 98). Some 30% of beef cattle are derived from calves from dairy herds and, of these, 45% are estimated to be finished under this system (Williams et al., 2006). We have assumed that the calves are autumn born, are housed for two winters and fed on silage and concentrates. Costs associated with these feedstocks are included in the analysis. In the summer, cattle graze grass fertilized with manufactured N.

Beef production profit expressed on a £ ha^{-1} basis continues to rise almost linearly with input (Figure 2a), but the TFP declines. The index is barely above 1, although excluding the environmental costs would raise the value of the index somewhat above one (data not shown). Figure 2b suggests that GHG emissions increase sharply with inputs in this system, the largest components of which are the N_2O emissions from denitrification of N fertiliser applied to the growing grass and feed, and the enteric fermentation to CH_4 during the growth of the animals themselves. These are large at all levels of production and increase with the intensity of production. Unlike arable systems, intensification in the stocking density does not lead to a reduction in the burden of land-use. This is because the animals eat more food than can be produced on the land used to raise them. These ‘external hectares’ increase more than the amount that the land area housing the animals decreases. We assume a constant yield for silage and for concentrates and have not attempted to map a variation in intensity of production in this part of the system onto the main beef production calculations.

3.2.2. Sheep meat

Production costs and values of output in the production of sheep meat are based on Nix (2005). In consultation with North Wyke Research (David Scholefield, personal communication) we have treated sheep in a similar fashion to beef since both are ruminant systems, but the intensity of production of finishing lambs is somewhat less. As with beef production we concentrated on a particular system known as 'grass grazed finished store lambs', which are grazed for 3 months on lowland grass. See supplementary material for a more detailed description.

The TFP index declines with input in the production of lambs (Figure 2c) even though profitability continues to rise. However, the scale is small (right-hand y axis) and it is difficult to elicit a real response to changes in input in this already low-input system. The environmental costs of lamb production are the least of all the systems we studied.

3.3 Production expressed on a per tonne basis

So far we have expressed costs and returns on a per hectare basis and we have taken the physiologically optimum yield as the reference point for our analysis. When inputs are reduced and yields are lower, we add the cost of using extra land to make up for the lost production. In this way, we have focused on the efficiency of systems that maintain current production rates.

If the breakdown in environmental costs is calculated on the basis of tonnes of product (Figure 3) the results for the arable crops remain much the same as on a per hectare basis. The minimum exploitation of the environmental resource occurs close to high intensities of production. This is true of lamb production too, but it is interesting to note that there is a minimum in the environmental costs associated with grazed beef that did not show up clearly where the results were expressed on a per hectare basis. In both animal systems, there is a trade-off between consumption of land and the emission of GHGs (Figs 3d and e), but in the beef system GHG emissions increase more and land consumption decreases less with intensity of production than is the case with sheep production. In the arable systems, the emission of GHGs and nutrient loss per tonne of product are reasonably constant across all levels of production, but pesticide pollution and land use increase at the lower levels of production (Figs 3a, b and c). These results have been related to consistent but different measures of intensity on the x-axes of Figure 3. Both high and low intensity production can give the same total cost (x-axes on Figures 1 and 2) when expressed on a per tonne basis, making the graphs difficult to read and interpret. Accordingly we have expressed intensity on the x-axis in non-monetary units.

4. Discussion

The relative contribution of the environmental burdens to agriculture, in financial terms, is interesting and surprising. Our analysis suggests that land-use and GHG emission are the most significant factors that determine system-wide sustainability (i.e. TFP > 1.0). The total GHG emission from the manufacture of all chemical interventions and farm operations are greatest at the most intense rates of production, and comprise the most significant environmental burden. Costs resulting from the emission of N_2O range from about £10 to £30 ha^{-1} moving from the least to the most intensive cropping systems. In animal production the figures are about £50 in lamb production to more than £200 in beef. This is a significant part of the total GHG emission from wheat, OSR and ruminant finishing systems, but the majority of the GHG burden associated with potatoes is in the lifting and storage of the tubers. The issues related to biocidal emissions do not change greatly with input, partly because we continue to apply insecticides and nematicides at the same rate per hectare to all levels of production. The loss of chemical inputs such as pesticides, is among the largest burdens at intermediate and high levels of production. At low levels of production, land consumption is the greatest issue in winter wheat and OSR production but land is less of an issue in finishing ruminants; for potatoes pesticide use and GHG production (chemical manufacture and harvesting and storage) are bigger concerns. Above the physiological maximum of crop production, N and P leaching and N_2O emissions increase and leaching begins to become more serious, particularly for potatoes. Note that the increase in the consumption of land becomes negative at high levels of intensity (Figs 1b, d & f) because, despite the fact that the optimum has been exceeded, production per unit area increases until *maximum* yield is achieved. The total environmental costs must reflect the fact that land is now producing slightly more per unit area in response to increased application of nitrogen.

4.1. Availability of data, uncertainties and assumptions

For arable production, the availability of data was good, mainly because arable cropping is a single-stage production system where the response to inputs is clear. Nutrient losses have been studied extensively during the last 15-20 years and, although the data cannot represent the detail of production in all parts of the UK, they nonetheless represent state of the art estimates at the national level. We have reasonable confidence in the way we have tied measurements of loss during field-based production with the national estimates of pollution and burdens provided by Pretty et al. (2005b 2003, 2000) and others (Atkinson et al., 2004). There are, however, differences in the values calculated by these authors for the environmental costs of different burdens, indicating differences of opinion as to the eventual future cost of pollutants emitted now. In all systems, the mapping of national levels of the costs of removing pesticides from drinking water, or of the burden of these chemicals to the environment, was difficult and must be considered highly uncertain. In general, Williams et al. (2006) suggest a variability of around 30% (CV) in national inventories and surveys, rising to 70% in the case of N_2O. Variability in farm inputs was thought to be <35%. The numbers we report are dependent on the assumptions made, usually to reflect average yields or a standard practice; inevitably there could be considerable variation about these averages and standards. These uncertainties will apply to the absolute value of the TFP index but we can have more confidence in the trends. Thus, while it may be difficult to pronounce this or that practice as sustainable in absolute terms, we believe that where we show significant changes in TFP with inputs we have captured real trends.

4.2 Environmental costs

At current values, it may seem surprising that the environmental costs are not a greater proportion of the whole. In part, this may be due to costs we have been unable to evaluate, such as the subjective cost of landscape or of the cost to ecosystems off-farm. It is also true that there is considerable uncertainty attached to the estimates of the environmental costs. However, if these values or the costs attributable to farming become available, our spreadsheets could be modified to take account of them. In several systems, particularly arable farming, it is the increase in land area needed to match national production levels that offsets any gain from reducing the intensity of production. Our estimates of the ecosystem services provided by land are conservative and derive from a 10-year old report that was itself conservative. Land would have to be valued at a much lower level before other environmental costs become significant enough to push the maximum in the TFP to lower levels of intensity of production. At much lower levels of production, economies of scale might decline and still more environmentally valuable land such as forest or natural ecosystems might be needed.

4.3 Multi-functionality

Espinosa et al. (2008) and Özkaynak et al. (2004) strongly emphasise the context of measures of sustainability. Our analysis is chiefly unimodal, although we have included the potential value of wheat straw (as bedding or biofuel, for example Powlson et al., 2008). We do not consider whole-farm TFP here, although this clearly would have an impact on decision making at the enterprise scale. Analysis of rotations is beyond the scope of this article but is clearly a topic worth further investigation. Indeed the relatively low value of TFP in OSR suggests that this crop might be grown partly for its benefit as a break crop to a following wheat.

Land can have more than one function and, if it is possible to promote a means to realise the value conferred on farm land by dealing with floods or providing a habitat for wildlife as well as growing a crop or finishing animals, then extensification might seem a more valuable course of action. Some of these qualities were included in the analysis of ecosystem services carried out by Costanza et al. (1997) but these authors did not consider arable land suitable for water capture, storage or regulation. Intercropping (either in space or time) might also raise the value of the sum of the outputs, the diversity of species in the land as well as reducing pollution (Whitmore and Schröder, 2007). It is also possible for improvements in the state of the system to have more than one benefit. For example, increased levels of organic matter not only increase fertility (Whitmore and Schröder, 1996) but also reduce the effort needed to plough (Watts et al., 2006). Furthermore, the source of the extra carbon is the atmosphere thus reducing the potential for global warming.

4.4 Temporality

A systems level definition of sustainability is that we should leave opportunities to the next generation equal or greater in value to those we enjoy. We have not explicitly considered the change in TFP over time in this analysis and have kept the costs of products and burdens static. To explore the dynamics of TFP as well as the effect of the rate of change of multiple inputs would have been unduly complex. The utility of the methodology presented here is its simplicity in the use of average values to capture the general balance between the competing components that determine whether or not a practice is sustainable. Clearly some information is lost in this way. In a theoretical analysis Cabezas and Fath (2002) elegantly express sustainability in terms of Shannon entropy or Fisher information, I. A process is sustainable if I is constant. If I declines this indicates that the system is becoming less sustainable, if I increases this indicates self-organisation. To estimate I requires detailed knowledge of the dynamics, which is beyond the scope of the relatively simple yet extensive analysis presented here.

Balmford et al (2002) objected to Costanza's economic valuation of all that is in planet earth on the grounds that the demand curve is unlikely to be linear and so as nature disappears, its value is likely to increase. Likewise the cost of food might increase disproportionately if it became scarce. In focussing on what will happen with fairly small shifts in production (±20% say as here) our assumption of a proportionate change cost is probably reasonable. It is clear, however, that strong pressures exist at the extremes and these will come into play if production is curbed or intensified greatly. Barnett et al. (1994) illustrate this with reference to the long-term experiment on winter wheat on Broadbalk field at Rothamsted and at Woburn. The index illustrates the differences in sustainability in the early years of the 20th Century and justifies the decision at that time to stop the experiment at Woburn while continuing the one at Rothamsted. Business failure, however, is not always about one year’s bad results. Lien et al (2007) following Hansen et al. (1997) derive the relative frequency of profitable years in order to test the sustainability of farming in the face of fluctuating conditions. In general, our analysis here has not attempted to take account of major changes or fluctuations in the cost or value of the components of our TFP index. Most obviously, if food is scarce its cost will increase. Less obviously, however, if land becomes damaged, production will fall, leading to a scarcity in food or if prices vary widely, it becomes difficult to plan season-long activities such as farming.

5. Conclusions

The intensity of the agricultural systems studied here that are optimal for production appears to be close to that which is optimal for the environment too, provided no loss of ecosystem service or productivity occurs in the land. Indeed wheat and OSR appear to be close to the *minimum* environmental burden level in current UK systems. In contrast to arable farming, ruminant finishing systems are characterised by increasing environmental exploitation with intensity of production (mainly nitrogen fertiliser use here) when expressed on *a per hectare* basis but there is a minimum in the environmental costs of all systems when expressed on *per tonne* basis. These minima are close to the actual intensities of production adopted by farmers in the UK.

At the time of writing, all systems investigated relied on support mechanisms to make them economically viable; recent increases in the value of arable crops and steep increases in the cost of oil may have changed the relationship between economic and environmental optima.

Attempting to manage any one or any of several environmental burdens such as GHG emission without reference to all, especially land, is likely to lead to an increase in exploitation of the unmanaged burden or to unintended results. Land area should be included in any system of environmental management. Introducing environmental incentives intended to reduce emissions without due reference to land, may have the result of pushing up land-use, land prices or both.

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Supplementary information

1. Appendices for arable, beef and sheep meat production (lodged with the journal for publication)
2. Spreadsheets for arable production (at www.rothamsted.bbsrc.ac.uk/aen/TFP/)

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Legend to Figures

Figure 1 *Total Factor Productivity (dashed lines) and yield response (solid lines) as a function of total costs ha^{-1}, including environmental costs (a, c, e) and breakdown of the environmental costs ha^{-1} as a function of total costs (b, d, f) for wheat (a, b), OSR (c, d) and potato production (e, f). See Methods section, Table 1 and supplementary information. Environmental burdens are as follows:*

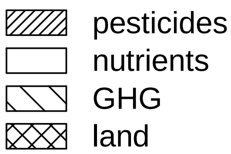

Figure 2 *Total Factor Productivity (dashed lines) and yield revenue (solid lines) as a function of total costs ha^{-1}, including environmental (a, c) and breakdown of the environmental costs ha^{-1} as a function of total costs (b, d) for beef (a, b) and lamb meat production (c, d). See Methods section, Table 1 and supplementary information. Environmental burdens as given in the legend to Figure 1*

Figure 3 *Breakdown of the environmental costs tonne^{-1} wheat (a), OSR (b) potatoes (c), beef (d) and sheep meat produced (e).). Loss of ecosystem services resulting from conversion of forest to agricultural use is fully costed. See Methods section, Table 1 and supplementary information Environmental burdens as given in the legend to Figure 1.*