

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

M.J. Glendining^a, A.G. Dailey^a, A.G. Williams^b, F. K. van Evert^c, K.W.T. Goulding^a and A.P. Whitmore^{a*}

^a*Centre for Soils and Ecosystem Function,*

Cross Institute Programme for Soil Sustainable Function (SoilCIP),

Department of Soil Science,

Rothamsted Research, Harpenden, Hertfordshire. AL5 2JQ. UK

^b*Natural Resources Department ,Cranfield University, Bedford.*

^c*Plant Research International, PO Box 16, NL-6700 AA Wageningen The Netherlands.*

*Corresponding author: andy.whitmore@bbsrc.ac.uk

Tel +44 (0)1582 763133

Fax +44 (0)1582 469036

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

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

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

53

54 **Keywords**

55 Sustainable Agriculture, Total factor productivity, Environment, Environmental

56 burden, Resource use, Environmental economics, arable, ruminant, Life Cycle

57 Assessment, Ecosystem services.

58

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.

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

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

208 Besides the economic cost and environmental burden of producing these chemicals
209 (primary energy, etc.), their use is itself an environmental burden. We have estimated
210 this burden as the sum of the costs of removing the compounds from drinking water,
211 costs to farmers and the National Health Service of acute damage to human health,
212 and the cost of the loss of abundance and diversity of wildlife. The costs of pesticides
213 to human health are thought to have been considerably underestimated as they do not
214 include chronic effects (e.g. cancers) and acute effects may well be under reported
215 (Pretty et al. 2000). In contrast, however, Trewavas (2004) avers that exposure to
216 manufactured pesticides and sprays is associated with lower rates of cancer than in the
217 general population. Notwithstanding this debate, Pretty et al. (2005a and 2000)
218 estimate environmental costs of chemicals for the whole of the UK. We use their data,
219 expressing them per hectare or per kg commodity by attributing the UK pesticide
220 costs first of all to commodities based on their relative production rates and on the
221 make up of a typical range of sprays used with each commodity, as explained in detail
222 in the appendix. Based on experimental results, cereal yields given by the Wheat
223 Disease Manager (Audsley et al. 2005) improve if sufficient amounts of bioicidal
224 chemicals of the correct kind are applied. We have chosen to invert this relationship
225 and so have derived the response at reduced applications. The national burden can
226 then be partitioned to crops at different rates of input. Chemicals are assumed to be
227 applied even if fertilisers are not. Reductions in chemical inputs are obtained by
228 reducing the number of sprays and accepting some actual reduction or risk of
229 reduction in crop yield from weeds, pests and diseases. Reducing the concentration of
230 active ingredient in a spray is not recommended because of the danger that the target
231 will develop resistance. We have not, therefore, used reduced concentrations in our
232 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 CO₂, methane CH₄ and nitrous oxide, N₂O 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 CO₂ equivalents, because the reference gas, CO₂ 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 today's 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 N₂O emissions from denitrification of N fertiliser applied to the growing grass and feed, and the enteric

fermentation to CH₄ 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

432

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

438

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

456

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 N₂O 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 N₂O 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 N₂O. 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

532 included the potential value of wheat straw (as bedding or biofuel, for example
533 Powlson et al., 2008). We do not consider whole-farm TFP here, although this clearly
534 would have an impact on decision making at the enterprise scale. Analysis of
535 rotations is beyond the scope of this article but is clearly a topic worth further
536 investigation. Indeed the relatively low value of TFP in OSR suggests that this crop
537 might be grown partly for its benefit as a break crop to a following wheat.

538
539 Land can have more than one function and, if it is possible to promote a means to
540 realise the value conferred on farm land by dealing with floods or providing a habitat
541 for wildlife as well as growing a crop or finishing animals, then extensification might
542 seem a more valuable course of action. Some of these qualities were included in the
543 analysis of ecosystem services carried out by Costanza et al. (1997) but these authors
544 did not consider arable land suitable for water capture, storage or regulation.

545 Intercropping (either in space or time) might also raise the value of the sum of the
546 outputs, the diversity of species in the land as well as reducing pollution (Whitmore
547 and Schröder, 2007). It is also possible for improvements in the state of the system to
548 have more than one benefit. For example, increased levels of organic matter not only
549 increase fertility (Whitmore and Schröder, 1996) but also reduce the effort needed to
550 plough (Watts et al., 2006). Furthermore, the source of the extra carbon is the
551 atmosphere thus reducing the potential for global warming.

553 *4.4 Temporality*

554
555 A systems level definition of sustainability is that we should leave opportunities to the
556 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 ($\pm 20\%$ 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.

607

608 Attempting to manage any one or any of several environmental burdens such as GHG

609 emission without reference to all, especially land, is likely to lead to an increase in

610 exploitation of the unmanaged burden or to unintended results. Land area should be

611 included in any system of environmental management. Introducing environmental

612 incentives intended to reduce emissions without due reference to land, may have the

613 result of pushing up land-use, land prices or both.

614

615

Acknowledgements

Rothamsted Research (RRes) is grant-aided by the UK Biology and Biotechnological Sciences Research Council. This project was supported by the GB Department of the Environment, Food and Rural Affairs (Defra, IS0216), and by the Dutch Ministry of Agriculture, Nature and Food Quality, project BO-5-398-I 331.010.3600.

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/)

References

- ABC. 2005. The Agricultural Budgeting and Costing Book No. 60. Agro Business Consultants Ltd, Melton Mowbray, UK.
- Addiscott, T.M., Whitmore, A.P., Powlson, D.S. 1991. Farming fertilizers and the nitrate problem. CAB International, pp 170.
- Atkinson, G., Baldock, D., Bowyer, C., Newcombe, J., Ozdemiroglu, E., Pearce, D., Provins, A. 2004. Framework for environmental accounts for agriculture. pp 105. Defra.
- Audsley, E., Milne, A.E., Paveley, N.D. 2005. A foliar disease model for use in wheat disease management decision support systems, *Ann. Appl. Biol.*, 147, 161–172.
- Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R.E., Jenkins, M., Jefferiss, P., Jessamy, V., Madden, J., Munro, K., Myers, N., Naeem, S., Paavola, J., Rayment, M., Rosendo, S., Roughgarden, J., Trumper, K., Turner, R.K. 2002. Economic reasons for conserving wild nature. *Science* 297, 950-953.
- Barnett, V., Johnston, A.E., Landau, S., Payne, R.W., Welham, S.J., Rayner, A.I. 1995. Sustainability – the Rothamsted experience. In: V. Barnett, R. Payne and R. Steiner (eds) *Agricultural Sustainability Economic, Environmental and Statistical considerations*. John Wiley, Chichester, pp171-206.
- Barnett, V., Landau, S., Welham, S.J. 1994. Measuring Sustainability. *Environmental and Ecological Statistics* 1, 21-36
- Cabezas, H. Fath, B.D. 2002. Towards a theory of sustainable systems. *Fluid Phase Equilibria* 194-197, 3-14.

652 Clarkson, R. Deyes, K. 2002. Estimating the Social Cost of Carbon Emissions, GES
 653 Working Paper 140, London: HM Treasury. Available at [www.hm-](http://www.hm-treasury.gov.uk/documents/taxation)
 654 [treasury.gov.uk/documents/taxation](http://www.hm-treasury.gov.uk/documents/taxation). pp 57.
 655 Costanza, R., D'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg,
 656 K., Naeem, A., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den
 657 Belt, M. 1997. The value of the world's ecosystem services and natural
 658 capital. *Nature* 387 253-260.
 659 Defra 2006. The British Survey of Fertiliser Practice. Fertiliser use of farm crops for
 660 the crop year 2005. London.
 661 Ehui, S.K., Spenser, D.S.C. 1992. A general approach for evaluating the economic
 662 viability and sustainability of tropical cropping systems: In M. Bellamy and B.
 663 Greenshields (eds.) *Issues in Agricultural Development*. Occasional paper of
 664 the IAAE. Oxford. pp 110-119.
 665 Espinosa, A., Harnden, R., Walker, J. 2008. A complexity approach to sustainability –
 666 Stafford Beer revisited. *European Journal of Operational Research*. 187, 636-
 667 651.
 668 Goulding, K.W.T., Jarvis, S.C. and Whitmore, A.P. 2008. Optimising nutrient
 669 management for farm systems. *Philosophical Transactions of the Royal*
 670 *Society series B* **363**, 667-680. doi:10.1098/rstb.2007.2177
 671 Hansen, J.W., Knapp, E.B., Jones, J.W. 1997. Determinants of sustainability of a
 672 Columbian hillside farm. *Experimental Agriculture*. 33, 425-448.
 673 Hartridge, O., Pearce, D. 2001. Is UK Agriculture Sustainable? Environmentally
 674 adjusted economic accounts for UK agriculture. CSERGE, University College,
 675 London

676 Jordan, V.W.L. 1991. Long Ashton low input farming and environment (LA. LIFE)..
 677 Schweizerische Landwirtschaftliche Forschung 29, 389-394.
 678 Jordan, V. W. L. 1992. Effect of low inputs of agrochemicals on cereal yields and
 679 profitability.. Bulletin of the Scottish Society for Crops Research 9, 16-19.
 680 Jordan, V. W. L. 1998. Less intensive integrated arable crop production and
 681 environmental protection. Journal of the Royal Agricultural Society of
 682 England 159, 56-69.
 683 Jordan, V. W. L., Hutcheon, J. A. 1993. Less-intensive integrated farming systems for
 684 arable crop production and environmental protection. Proceedings of the
 685 Fertiliser Society 346, 32pp.
 686 Lien, G., Hardaker, J.B., Flaten, O. 2007. Risk and economic sustainability of crop
 687 farming systems. Agricultural Systems. 94, 541-552.
 688 Lynam, J.K. Herdt, R.W. 1989. Sense and sensibility: sustainability as an objective in
 689 international agricultural research. Agricultural Economics 3, 381-398.
 690 MAFF 2000. Fertiliser recommendations for agricultural and horticultural crops
 691 (RB209) 7th Edition, pp 177.
 692 Nix, J. 2005. Farm Management Pocketbook, 35th Edition. Imperial College London
 693 Wye Campus.
 694 Özkaynak, B., Devine, P., Rigby, D. 2004. Operationalising strong sustainability:
 695 definitions, methodologies and outcomes. Environmental Values. 13, 297-303
 696 Powlson, D.S., Riche, A.B., Coleman, K., Glendining, M.J., Whitmore, A.P. 2008.
 697 Carbon sequestration in European soils through straw incorporation:
 698 limitations and alternatives. Biowastes. 28, 741-746.
 699 doi:10.1016./j.wasman.2007.09.024

700 Pretty, J. N., Ball, A. S., Lang, T., Morison, J. I. L. 2005a. Farm costs and food miles:
 701 An assessment of the full cost of the UK weekly food basket. Food Policy 30,
 702 1-20.

703 Pretty, J.N., Brett, C., Gee, D., Hine, R.E., Mason, C.F., Morison, J.I.L., Raven, H.,
 704 Rayment, D., van der Bijl, G. 2000. An assessment of the total external costs
 705 of UK agriculture. Agricultural Systems. 65, 113-136.

706 Pretty, J.N., Heffron, L., Nedwell, D.B. 2005b. The costs and benefits of nitrogen
 707 enrichment. *In* The NERC Global Nitrogen Enrichment (GANE) programme
 708 finale, February 16th 2005.
 709 <http://www.nerc.ac.uk/funding/thematics/gane/documents/pretty.pdf>, Royal
 710 Geographical Society, London, 2005b.

711 Pretty, J.N., Mason, C.F., Nedwell, D.B., Hine, R.E., Leaf, S., Dils, R. 2003.
 712 Environmental costs of freshwater eutrophication in England and Wales.
 713 Environmental Science & Technology 37, 201-208

714 Scholefield, D., Lockyer, D.R., Whitehead, D.C. and Tyson, K.C. 1991. A model to
 715 predict transformations and losses of nitrogen in UK pastures grazed by beef-
 716 cattle. Plant and Soil 132, 165-177.

717 ten Berge, H.F.M., Withagen, J.C.M., de Ruiter, F.J., Jansen, M.J.W., van der Meer,
 718 H.G. 2000. Nitrogen responses in grass and selected field crops. Quad-mod
 719 parameterisation and extensions for STONE-application. Report 24 Plant
 720 Research International, pp 42 plus appendices.

721 Trewavas, A. (2004) A critical assessment of organic farming-and-food assertions
 722 with particular respect to the UK and the potential environmental benefits of
 723 no-till agriculture. Crop Protection 23, 757–781.

724 Watts, C.W., Clark, L.J., Poulton, P.R., Powlson, D.S., Whitmore, A.P. 2006. The
725 role of clay, organic carbon and cropping on plough draught measured on the
726 Broadbalk Wheat Experiment at Rothamsted. *Soil Use & Management*. 22,
727 334-341.

728 Whitmore, A.P., Schröder, J.J. 2007. Intercropping reduces nitrate leaching from under
729 field crops without loss of yield: A modelling study. *European Journal of*
730 *Agronomy*. 27, 81-88. doi:10.1016/j.eja.2007.02.004

731 Whitmore, A.P., Schröder, J.J. 1996. Modelling the change in soil organic C and N in
732 response to applications of slurry manure. *Plant and Soil* 184, 185-194.

733 Williams, A.G., Audsley, E., Sandars, D.L. 2006a. *Determining the environmental*
734 *burdens and resource use in the production of agricultural and horticultural*
735 *commodities*. Main Report. Defra Research Project IS0205. Bedford:
736 Cranfield University and Defra. Available on www.agrilca.org, and
737 www.defra.gov.uk

738 Williams, A.G., Audsley, E., Sandars, D.L. 2006b. Energy and environmental burdens
739 of organic and non-organic agriculture and horticulture. In: What will organic
740 farming deliver? COR 2006, Eds. Atkinson, C. et al. Association of Applied
741 Biologists, Wellesbourne, Warwick CV35 9EF, UK:, 19-24.

742

743

744

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:

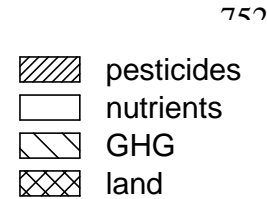


Figure 2 Total Factor Productivity (dashed lines) and yield revenue (solid lines) as a function of total costs ha^{-1} , including environmental (a, c) and breakdown of the environmental costs ha^{-1} as a function of total costs (b, d) for beef (a, b) and lamb meat production (c, d). See Methods section,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

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

Glendining, Margaret J.

2009-02-01

M.J. Glendining, A.G. Dailey, A.G. Williams, F.K. van Evert, K.W.T. Goulding, A.P. Whitmore, Is it possible to increase the sustainability of arable and ruminant agriculture by reducing inputs?, *Agricultural Systems*, Volume 99, Issues 2-3, February 2009, Pages 117-125

<http://dx.doi.org/10.1016/j.agry.2008.11.001>

Downloaded from CERES Research Repository, Cranfield University