IS UK AGRICULTURE SUSTAINABLE? ENVIRONMENTALLY ADJUSTED ECONOMIC ACCOUNTS FOR UK AGRICULTURE

Olivia Hartridge
David Pearce

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CSERGE-Economics
University College London

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At the time of writing the early drafts of the paper Olivia Hartridge was at CSERGE-Economics at University College London. She is currently with the Environment Protection Economics division at the UK Department for the Environment, Food and Rural Affairs (DEFRA). The views expressed in this article are the authors' own and must in no way be taken to reflect any viewpoint on the part of HM Government.

Abstract

Agricultural sectors in most advanced economies have come under severe criticism for lacking the characteristics of 'sustainability'. What is usually meant is that a combination of subsidies and modern farming methods is producing an economically and environmentally non-viable agricultural sector. Using economic valuation techniques, and adjusting for prevailing subsidies, we seek to re-estimate the contribution that the agricultural sector made to the UK economy in the year 1998. The sector is markedly smaller if adjustments are made for subsidies. But these subsidies allow the sector to be a generator of both substantial environmental benefits, and also of extensive environmental damages.

1 Introduction

For an economy to be sustainable, it must provide non-decreasing per capita welfare over time. Numerous efforts have been made to establish the conditions for the achievement of this notion of sustainability, and some of these have resulted in indicators and measures of sustainability (for a review see Atkinson et al, 1997). In one form or another, these measures and indicators try to modify normal measures of economic activity, such as Gross National Product (GNP), by allowing for environmental impacts in the form of depleted resources and pollution damage. One economic sector of concern in this respect has been agriculture in developed economies since (a) it is extremely land intensive and hence environmentally important, and (b) it is the beneficiary of widespread and costly subsidies. These concerns have motivated a drive for 'sustainable agriculture' in which modern farming methods are modified to adopt low-input technologies and the sector is contracted to survive without subsidies.

In this paper we attempt to measure the sustainability or otherwise of the UK agricultural sector by adjusting Net Domestic Product for the year 1998. Partial exercises for other countries can be found in Tiezzi (1999) for Italy, Bonnieux et al. 1998 for France, le Goffe (2000) for France, and Hrubovcak et al. (2000), Smith (1992) and Steiner et al. (1995) for the USA. The only other studies relating to the UK are Adger and Whitby (1991, 1993) and Pretty et al. (2000). We compare our estimates to these latter studies. We find that negative externalities amount to at least £1 billion, whilst positive externalities (the amenity value of the agricultural countryside, excluding non-use values) offsets approximately one half of these negative effects. These externalities relate to the sector as it is currently. If subsidies were removed, the sector would contract and the configuration of externalities would differ. Accordingly, it is not legitimate to add our estimate of externalities to the 'commercial' valued-added of the sector when adjustments are made for subsidies. We therefore present two adjustments: (a) the probable NDP of the sector without subsidies, and (b) the externalities associated with the existing 'with subsidy' situation. We later speculate on what the overall measure of modified agricultural NDP might be.

2 Environmental Accounting

From the many attempts to develop modified economic accounts, this paper adopts the approach taken by Hamilton and Atkinson (1995) which develops the earlier 'genuine savings' approach introduced by Pearce and Atkinson (1992). At the core of environmental accounting is the Hicksian definition of income (after Hicks, 1939). This suggests that the 'true' measure of income is the level of income that can be consumed whilst maintaining intact the stock of capital assets over the accounting period. Weitzman (1976) showed that, for a competitive economy following an optimal growth path, Net Domestic Product is the constant level of output, the present value of which equals the present value of consumption. Following Hamilton and Atkinson (1995) this result can be expressed formally as follows, focusing initially on the degradation of environmental services. The second type of depreciation - depletion of resources - is incorporated later.

Utility, or wellbeing, is derived from the consumption of income C, and the level of environmental services B:

$$U(C, B) \quad \ldots\ldots[1]$$

Assume that this economy produces a single, composite good, which can either be consumed or invested. The resulting gross domestic product (GDP) has the following production function:

$$F(K,L) = C + \dot{K} \quad \ldots\ldots[2]$$

where K is man-made capital, L is (an unconstrained) supply of labour, and the dot above any variable denotes rate of change. Pollution e is emitted as a by-product of the production process e(F). This flow of pollution contributes to a stock M. Environmental services are negatively related to this stock.

$$\begin{aligned} \dot{M} &= e \\ B &= B_O - \alpha M \end{aligned} \quad \ldots\ldots[3]$$

α is a parameter quantifying the impact a unit of pollution has on the level of environmental services.

Utility is maximised subject to the impact of consumption on the man-made capital stock and the impact of production on the level of environmental services.

$$\begin{aligned} &\max \int_O^{\infty} U(C,B) e^{-rt} dt \\ &s.t. \quad \dot{K} = F - C \qquad \ldots\ldots[4] \\ &\qquad\quad \dot{M} = e \end{aligned}$$

The current value Hamiltonian is:

$$\begin{aligned} \tilde{H} &= U(C,B) + \gamma_1 \dot{K} + \gamma_2 \dot{M} \\ &= U(C,B) + \gamma_1 (F - C) + \gamma_2 e \end{aligned} \quad \ldots\ldots[5]$$

with shadow values γ_1 and γ_2. Maximising with respect to consumption, the control variable, we have:

$$\frac{d\tilde{H}}{dC} = 0 = U_C - \gamma_1 \quad \Rightarrow \quad U_C = \gamma_1 \ \ldots\ldots[6]$$

Thus the shadow price of capital, in units of utility, is the marginal utility of consumption. The Hamiltonian can be used to find the shadow prices for pollution and environmental services, in units of utility. If these shadow values are then divided by the marginal utility of consumption, a measure of economic welfare (MEW) can be expressed using these shadow values in consumption units:

$$MEW = C + \dot{K} + \frac{\gamma_2}{U_C} e + \frac{U_B}{U_C} B \ \ldots\ldots[7]$$

where γ_2 is the shadow price of pollution in utility units, and, when converted into consumption units, $\sigma = -\gamma_2/U_C$ is the marginal social damage cost of pollution. σ is negative because as pollution increases, welfare is decreased. At the optimum, σ will equal the marginal abatement cost of pollution. Therefore, at the optimum, σ is equal to the rate of a Pigouvian tax, and damage from pollution will not be equal to zero. There will be an optimal level of the flow of pollution and an optimal stock level of pollution. It is not theoretically correct therefore to value pollution damage by estimating the cost of returning the environment to its condition prior to damage, the method that is sometimes resorted to in the literature.

U_B is the shadow price of environmental services, and when converted into consumption units is the maximum willingness to pay (WTP) p_B for environmental services. Thus, the components of natural capital fit the standard model of consumer preferences and utility maximisation, and σ and p_B are ratios of marginal utilities.

$$MEW = C + \dot{K} - se + p_B B \;\ldots\ldots[8]$$

This basic structure can be extended to include depletion r, and growth g, of the stock of renewable resources such as forests, and the natural dissipation d, of pollution emissions in the atmosphere - both extensions being relevant to the agricultural and forestry sector. (The depletion and discovery of non-renewables could also theoretically be included but are not directly relevant to this study). Hence,

$$MEW = C + \dot{K} - s\,(e - d) - n(r - g) + p_B B \;\ldots\ldots[9]$$

The shadow value n, of resource depletion and growth is price minus marginal cost: a Hotelling rent. Hotelling rents are used to value depletion of non-renewable resources in a theoretical argument close in character to the Hotelling Valuation Principle, and this can be extended to renewable resources. The price of the resource is assumed to rise at the rate of interest through time, implicitly assuming a constant marginal cost. Hence, the value of the total rent for the year, if left in the ground, would also rise at the rate of interest through time. Taking the present value means that depreciation of the asset can be estimated solely by taking the value of the total rent for the year thereby avoiding the use of a discount rate. The problem that remains is finding data for the marginal cost of production. In reality, the simplification to using average cost is often made. If, however, marginal costs are rising through time as depletion continues, using average costs as a proxy will mean that the aggregate Hotelling rents are overestimated, and hence the value of depletion to be deducted from GNP will be overestimated.

Adjusting net domestic product (NDP) for the UK agricultural and forestry sector also involves interpreting the model with reference to transboundary damage to natural capital. In particular the sector is responsible for contributing to the stock of greenhouse gases (GHGs) in the global atmosphere, as well as emitting regional air pollutants that contribute to acid rain in Europe. We assume the polluter-pays-principle holds and, in any event, the UK is a signatory to international agreements requiring GHG and acid rain reductions. Therefore the agricultural and forestry sector accounts will have environmental damage deducted from GDP for emissions affecting other countries, valued at a shadow price which adds in the marginal social damage cost to other countries.

Due to the interconnections within the domestic economy, it is not easy to decide what environmental damages an individual economic sector is responsible for, a problem that does not arise with nationally aggregated damages. We will assume that the agricultural and forestry sector is not responsible for any environmental damage caused during the manufacture of its inputs to production (which would be relevant for a full 'life cycle' approach). Thus, any harmful chemicals released into the environment during the manufacture of pesticides or fertilisers will not be attributed to agriculture or forestry. But, what will be attributed to the sector is the damage pesticides and fertilisers inflict on the environment once they have been purchased by the sector. In the same way, the sector is not held accountable for damage from emissions from power stations during the conversion from fossil fuels to energy (indirect emissions), even though the sector uses energy to power machines and buildings. The sector will be held responsible, though, for the emissions damage from fossil fuels that it uses on-site to power its tractors and other vehicles (direct emissions). Clearly, there are arguments for and against a full life-cycle approach but inclusion of all life cycle impacts that would occur because of the existence of the agricultural sector requires information that we consider is not yet available.

Most importantly, agriculture and forestry will be credited with the removal of carbon from the atmosphere via carbon fixing in biomass such as trees and soil, irrespective of the outcome of products leaving the sector. Therefore, timber products purchased from the sector may well be burnt and the carbon re-released in to the atmosphere, but the initial carbon credit will not ex post be debited. The credit will remain in the agricultural and forestry accounts, but will be debited accordingly from the purchaser’s sectoral accounts.

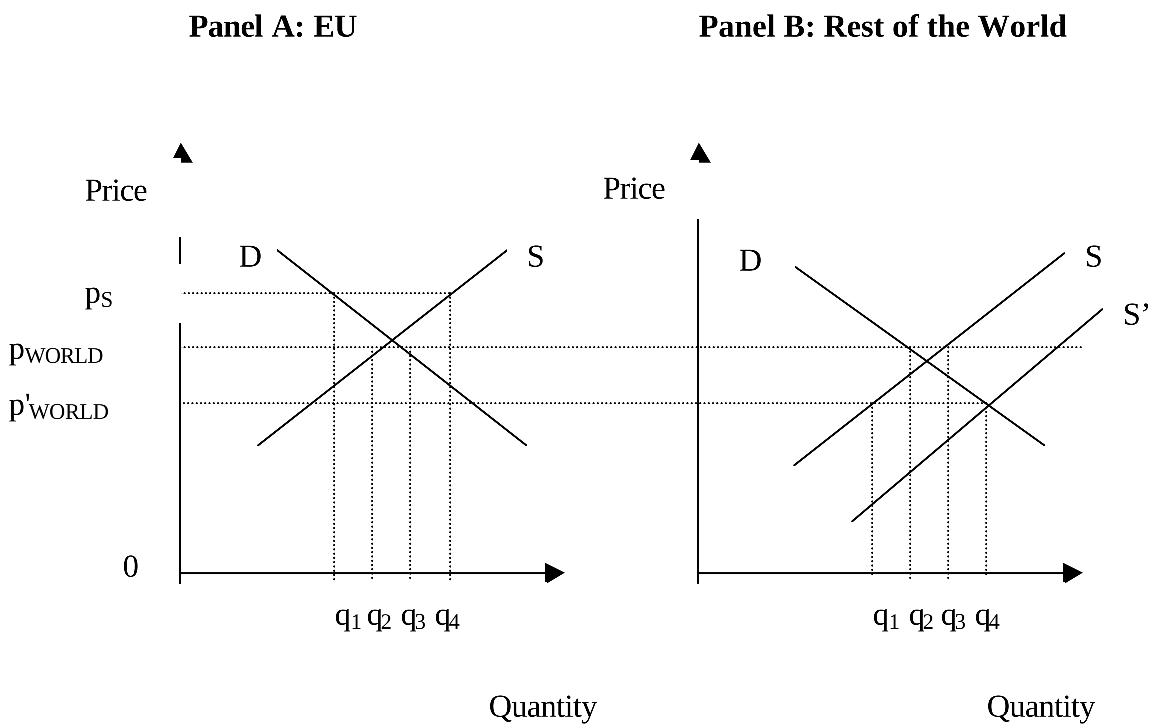
3 Correcting for subsidies

Aside from adjustments to agricultural accounts for environmental impacts, agriculture is heavily subsidised in the European Union. Hence, independently of any externalities, its output value ideally should be adjusted for the correct shadow prices. These are border or world prices since these reflect the opportunity cost of agricultural production in terms of what would have to imported if the sector did not produce for domestic consumption.

The problem is that the world prices that would prevail in the absence of subsidies are not easily estimated as a counterfactual, not least because they depend on what other countries would do in respect of any change in sectoral subsidy policy. There are additional problems. Whilst we can hypothetically use world prices to derive the first stage of adjusted accounts, and then modify these accounts for externalities, in practice the removal of subsidies would alter the nature of the farming sector. There may be increases in intensification in some areas whilst land price changes may encourage extensification in others. Therefore, in this paper, we first present the reduction in NDP when subsidies are removed (from basic to market prices), and then secondly modify NDP at basic prices for externalities. We recognise that current data does not allow us to re-evaluate NDP at world prices. Also, that either re-evaluation or deduction of subsidies would still not permit us to contribute valuations of externalities at subsidised prices. This area is clearly a subject for future development.

Formally, the situation is at set out in Figure 1 below. Panels A and B contain supply and demand curves for the European Union and the rest of the world respectively. D is the demand for the supported product in both markets. S is both the supply of the product before support has been implemented in the EU, and the supply that will not be supported in the rest of the world. P_{WORLD} is the resulting world market price, and the EU supplies q_2 and demands q_3. Hence the EC is a net importer of (q_3-q_2) from the rest of the world.

Figure 1: Effects of a price support on EU and world markets



Once a price support has been implemented at p_S, the EU begins to supply q_4 and to demand q_1 in response to a higher price. A surplus of (q_4-q_1) is generated which cannot be absorbed at the support price in the EU due to lack of domestic demand. It has to be exported to the rest of the world at world prices. This increases supply in the rest of the world to S', and lowers the world price to p'_{WORLD}. The rest of the world becomes a net importer, and the Common Agricultural Policy is left shouldering a financial burden the size of the rectangle $p_S p'_{WORLD} q_1 q_4$. Consumers in the EU pay an extra $p_S p'_{WORLD} 0 q_1$. These burdens increase and decrease as world prices fall and rise respectively. To summarise, EU farmers gain in welfare whilst EU consumers lose, and farmers in the rest of the world suffer a welfare loss whilst their consumers gain. The procedure adopted here is to value net domestic product (NDP) at p'_{WORLD}. There are insufficient data to evaluate NDP at the undistorted price, p_{WORLD}.

4 The Physical Accounts

The following physical accounts detail the depreciation of, and environmental services from the stock of natural capital associated with agriculture. Natural capital is divided into five resources: air, water, soil, landscape, and biodiversity. Air is further subdivided into global, regional and local categories.

The physical changes that occur to each resource due to sectoral activity are listed below in three parts:

1. The nature of the depreciation or appreciation of this component of natural capital;
2. The methodology used in measuring the extent of the depreciation or appreciation;
3. The problems involved with using this methodology.

4.1 Greenhouse gas emissions and global warming

The greenhouse gases identified in the Kyoto Protocol are carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), and sulphur hexafluoride (SF_6). Agricultural and forestry activity contributes to global emissions of the first three of these six gases.

The sector emits CO_2 in three main ways. Firstly, soil organic carbon is oxidised to CO_2 during cultivation of arable land or semi-natural vegetation, when the soil is rotated to the surface and exposed to the air. Cultivation also makes soil vulnerable to wind or water erosion which further oxidises the soil's carbon content whilst in the air or in surface waters. In the period 1947-1980, a net average of 18,000 ha per year was changed from semi-natural vegetation into agriculture, and from 1984 to 1990 land was converted back at a rate of 45,000 ha per year. However, from today's perspective, the land converted before 1980 and remaining in cultivation is still losing carbon, whilst the land which was converted back has only a slow rate of accumulation. This results in a net loss of carbon from the soil, and hence the release of CO_2. The second source of CO_2 is when peat or fenland is drained in readiness for the planting of commercial forests or arable crops. 254,000 ha of peat have been drained for forestry in the UK over the last few decades in order to aerate the top 60-70 cm of soil. Thirdly, CO_2 is released during the combustion of fossil fuels to power tractors and other vehicles.

But the agriculture and forestry sector also acts as a sink for CO_2 in three main ways. Firstly, carbon accumulates in biomass in plantation forests and litter, and on non-forest land. From 1920-1990, about 2 million ha of new forests were planted in Britain and these plantations are still absorbing carbon. Non-forest land comprises crops on arable land, land set-aside, woodland grown on farmland in response to the UK Farm Woodland Premium Scheme since the beginning of the scheme in 1992, and the conversion of rough grass to permanent grass. Secondly, carbon has accumulated in soil in afforested land, in land set-aside, and in undrained peatland. The third sectoral sink is a result of increasing atmospheric concentrations of nitrogen compounds (NO_X and NH_X), and CO_2. These gases act as fertilisers increasing the carbon stored in vegetation and soils. For example, some studies show evidence of increasing forest growth rates (see Worrell (1987), Tyler et al (1996), Cannell et al (1998)).

The loss of carbon from soil as a result of urbanisation has not been entered into these carbon accounts. This is because when land is sold from the agriculture and forestry sector to the urban sector it has not lost any carbon until developed by the urban sector. Thus the loss of carbon is charged to the urban sector. In the same way, the possible end uses of wood products sold by forestry are not included in the accounts. Any loss of carbon through wood products should be charged to the relevant purchaser's sector. Again, this applies to the storage of carbon in crops and livestock. Despite the fact that when those crops are sold to another sector and digested as food the carbon will be released, the carbon has still been fixed by the agricultural and forestry sector. However, this demarcation causes a problem with respect to the first sectoral sink mentioned above. The accumulation of biomass in non-forest land includes the minimal absorption of carbon through land converted from agriculture to urban uses. The total figure cannot be disaggregated to minus this entry which should be charged to the urban sector. Thus, the total figure is a small overestimate of the figure that should be charged to the agricultural and forestry sector.

Agriculture is the second largest source of methane gas in the UK after landfill sites. This is because methane is formed during the decomposition of organic matter. Hence livestock produce CH_4 from enteric fermentation and decomposition of animal wastes. Anaerobic conditions in wetlands also produce CH_4.

Table 1 Estimation of sources and sinks of CO_2, CH_4, and N_2O

CO_2 (Million tonnes C)

Sinks		Sources	
Forest biomass and litter	-2.1	Cultivation of soil	6.2
Non-forest biomass	-0.3	Drained peatland	0.3
Forest soils	-0.1	Drained fenlands	0.5
Set-aside soils	-0.4	Peat extraction	0.2
Undrained peatlands	-0.7	Export to sea of soil from rivers	1.4
CO_2 and N fertilisation	-2.0	Fossil fuel combustion	1.1
Crops for consumption	-6.0		
Livestock	-0.5		
Total	-12.1	Total	9.7

Note: negative figures denote appreciation of natural capital.
All data for carbon, except fossil fuel combustion and crop and livestock consumption, taken from Cannell et al (1999) which estimates carbon sources and sinks in 1990. Data for fossil fuel combustion communicated via e-mail from NETCEN (1997 figures). Data for crop and livestock consumption taken from Adger and Whitby (1993) (1988 figures).

CH_4 (Million tonnes CH_4)

Sinks		Sources	
		Enteric fermentation	0.94
		Animal wastes	0.124
		Soil	0.201
Total	-	Total	1.265

Note: DETR. Digest of Environmental Statistics 1998 (1996 figures), except for soil which is taken from Adger et al 1994 (1980 figures).

N_2O (Million tonnes N_2O)

Sinks		Sources	
		Animal wastes	0.0051
		Non-livestock agriculture	0.0932
Total	-	Total	0.0983

Nitrous oxide is formed from nitrogen fertilisers and from the treatment and disposal of animal wastes. Agriculture is the largest source of N_2O in the UK.

For carbon, the estimates were reached using a variety of methodologies. The model used to calculate changes in forest biomass recreates annual changes in forests such as tree growth using yield classes, harvesting, periodic thinning, and the decomposition of litter. In contrast the model used to estimate changes in non-forest biomass assumes equilibrium values for accumulated carbon in different land-use categories, and uses agricultural census statistics to give aggregate values. Other estimates such as soil carbon accumulation and oxidation resulted from modelling exponential losses and gains. The variety of methodologies reflects the difficulty in measuring changes. For soil in set-aside and cultivation, the uncertainty concerning the measurements is as high as $\pm 50\%$. Data for methane result from direct measurements from livestock, and this is reflected in the lower level of uncertainty ± 20-30%. The methodology for measuring nitrous oxide is not given in DETR (1998), nor is the level of uncertainty.

Table 1 summarises our estimates of emissions of greenhouse gases.

4.2 Regional: 'acid rain'

The compounds sulphur dioxide SO_2 and nitrogen oxides NO_X react with water molecules in the atmosphere to form acids which then dissolve into rainwater, this being known as 'acid rain'. Acidic depositions can however be both wet and dry. They constitute a regional externality because emissions are transported across national boundaries. The acid reduces agricultural crop yields, erodes building infrastructure, exacerbates human breathing difficulties, and increases mortality rates.

Emissions from the UK agricultural and forestry sector for SO_2 and NO_X are negligible (less than 1% and 2% of the total respectively) and are measured against a total that is decreasing due to international pressure. They are released from the sector’s combustion of fossil fuels.

SO_2 emissions are difficult to estimate since, except for large combustion plants, information is not available on actual emissions from specific, individual sources. Thus, estimates have to be made on the basis of emission factors for different fuels used by different sources, and combined with statistical information on patterns of use. NO_X emissions from vehicle exhausts are estimated using speed-related emission factors. The estimates have levels of uncertainty ± 10-15% and $\pm 30\%$ respectively.

Table 2 Regional deposition of SO_2 and NO_X due to UK agriculture

Million tonnes

Pollutant	Source	Total emissions	Emissions deposited regionally
SO_2	Fossil fuel combustion	0.0055	16% of 0.0055 = 0.00088
NO_X	Fossil fuel combustion	0.0297	27% of 0.0297 = 0.00802

Note: figures for local deposition of SO_2 and NO_X are estimated separately under local effects. Data for air pollutants emitted from fossil fuel combustion, SO_2, NO_X and PM_{10}, all communicated via e-mail from NETCEN (1997 figures). NO_X measured in NO_2 equivalent.

EMEP (EMEP, 2000) figures estimate that for SO_2, around 57% lands in the sea where it is assumed to do no damage, 27% is deposited locally in the UK and the remaining 16% is deposited regionally in other countries. The equivalent percentages for NO_X are 58%, 15% and 27% respectively.

4.3 Local air pollution

Airborne particulate matter is a diverse material in terms of its physical and chemical properties. The mass of material containing particles of up to ten micrograms in diameter is called PM_{10} which is known to exacerbate respiratory illness. Non-methane Volatile Organic Compounds (NMVOCs) cause damage via photochemical oxidation forming tropospheric ozone. The one exception is benzene, which is a known human carcinogen.

Emissions of PM_{10} from the agricultural and forestry sector are through the combustion of fossil fuels and so are again negligible, whilst NMVOC emissions mainly arise naturally from forests. NMVOC emissions from the sector are 8% of total emissions, but have reduced markedly since straw and stubble burning was banned in 1992.

Table 3 Local emissions of SO_2, NO_X, PM_{10} and NMVOCs

Million tonnes

Pollutant	Source	Emissions	Emissions deposited locally
SO_2	Fossil fuel combustion	0.0055	27% of 0.0055 = 0.00149
NO_X	Fossil fuel combustion	0.0297	15% of 0.0297 = 0.00446
PM_{10}	Fossil fuel combustion	0.0029	0.0029
NMVOCs	Forests	0.178	0.178

Note: emissions for NMVOCs obtained from Digest of Energy Statistics 1999 (1997 figures).

4.4 Water quality in rivers and canals

Water pollution problems stem from the sector’s use of fertilisers and pesticides, the increasing use of silage, and the quantities of slurry produced.

Plants absorb potassium (K), nitrogen (N) and phosphorus (P) up to the level of their needs. Any excessive nutrients in the soil from overuse of fertilisers and livestock manure find their way into water courses. This causes problems of eutrophication in river and coastal waters. The excess nutrients enable algae to accelerate growth, which in turn depletes the oxygen available to other living species, decreasing their population sizes.

The Environment Agency measures biological water quality in rivers and canals by monitoring macro-invertebrates living in or on the bed of rivers. The number and diversity of freshwater species in a river is compared to the species expected to be present in the absence of pollution. This is used to infer biological water quality. The problem is that the decrease in biological water quality cannot be entirely attributed to the agricultural and forestry sector. There are no data on direct emissions of nutrients from the sector: only an estimate that agriculture contributes two-thirds of all nitrogen emissions, and a third of all phosphate emissions into water courses as an EU average (OECD, 1998). These figures can only be used directly in the calculations for riverine discharges of nutrients (see below). A second problem is that river quality varies with the annual rainfall. The annual rainfall from 1995-1997 was below the long-term average for Britain, so the concentrations of pollutants would have been higher at the time of measurement, other things being equal.

Table 4 shows the state of water quality.

Table 4 Biological water quality in the UK for rivers and canals

Kilometres

Country	Good	Fair	Poor	Bad
England and Wales	24860	9930	2040	720
Scotland	14960	1360	280	110
Northern Ireland	1680	640	10	-
Total	41500	11930	2330	830

Source: Digest of Environmental Statistics (1995 figures for England, Wales, Scotland, 1996 figures for Northern Ireland).

4.5 Riverine discharges of nutrients in coastal areas

Nutrients not residing in inland waters are discharged into the sea in coastal areas. Again eutrophication can occur. The Environment Agency found estimates of discharges by sampling each main river system once per month approximately, at a sample point close to, but upstream, of the tidal limit. The survey aimed to monitor 90% of the discharges for each pollutant. As discussed above, the figures for nitrogen and phosphorus should be multiplied by two-thirds and a third respectively, according to OECD estimates of sectoral discharges. The OECD estimates are rough figures, but the error will be greater if they are not used.

Table 5 Riverine discharges of nutrients into coastal waters around the UK

Million tonnes

Nutrients	Lower	Upper
Orthophosphates	0.03	0.031
Total nitrogen	0.304	0.306
Nitrogen from ammonium	0.063	0.064
Total	0.397	0.401

Source: Digest of Environmental Statistics 1998 (1996 figures). Since some samples contained quantities of substances below detection limits, an upper and a lower estimate were devised. Lower: assume true concentration zero; upper: assume true concentration at limit of detection.

4.6 Riverine discharges of pesticides into coastal areas

Pesticides protect crops and plantations from competing species, but, due to their toxic nature, cause damage when released into water systems. This dissemination occurs in a number of different ways: direct application, spray drift, volatilisation, run-off, leaching and soil erosion. Damage to the environment includes changing the species' composition and developing pest resistance. Pesticides are also suspected of damaging the endocrine system in vertebrates, via mimicking or preventing the action of steroid hormones (Howarth et al 1999).

Increasing detection over the last few years of data is due in part to greater sensitivity of analytical techniques (DETR 1999), but low rainfall may also be an explanatory variable. Also, the figures in Table 6 for atrazine and simazine may be over-estimates for 1998 because they were both banned in 1993 and traces of these pesticides will have decreased below the 1995 figures given. In addition, other sectors use pesticides, for example, public sector services spray roadsides, so the figures over-estimate the damage inflicted by the agricultural and forestry sector to water systems.

Table 6 Riverine discharges of pesticides into coastal areas of the UK

(Tonnes)

Pesticides	Lower	Upper
Lindane	255	370
Atrazine (banned 1993)	0.55	2.32
Azinphos-ethyl	0.00	1.44
Azinphos-methyl	0.00	1.74
DDT	0.02	0.30
Dichloros	0.05	1.60
Drins	0.02	0.79
Endosulfan	0.00	0.26
Fenitrothian	0.00	1.28
Fenthian	0.00	1.11
Malathian	0.00	1.20
Parathian	0.00	0.34
Parathian-methly	0.00	1.21
Simazine (banned 1993)	1.22	2.57
Trifluralin	0.03	0.94
Total	256.89	387.1

Source: Department of the Environment, Transport and Regions (1998) (1995 figures).

This section deals solely with pesticide releases to water courses. It is widely suspected that pesticide spraying in general is implicated in significant biodiversity loss in the UK (Joint Nature Conservation Committee, 1997). If so, the damage from pesticides will be far greater than that accruing from watercourse concentrations. Sales of all pesticides in the UK (including fungicides and herbicides) are around 25,000 tonnes.

4.7 Water pollution incidents

Intensification of farming practices in the last twenty years has meant a greater number of animals being kept in pens on concrete rather than being kept outside or bedded on straw. It has also meant more being fed on a diet of silage rather than hay. The result has been a vast increase in the quantity of slurry and silage produced. Silage is produced via bacteriological breakdown of grass and other organic matter under pressure in silos or clamps (Joint Nature Conservancy Council, 1991). Slurry and silage effluent are up to 100 and 200 times respectively more polluting than untreated human sewage in terms of biochemical oxygen demand (BOD). Therefore, when yard washings, slurry spread on fields, and silo leakages reach water courses, the environmental damage can be significant.

Pollution incidents in aggregate have been decreasing over the 1990s, and the Environment Agency's 1998 annual report (1996 figures) recorded a further decrease. However, 1998 was a relatively wet year, after three relatively dry years, and correspondingly, the proportion of agricultural (organic waste) incidents increased. Hence the 1996 figures shown in Table 7 could be seen as fairly representative of 1998.

Table 7 Reported and substantiated water pollution incidents

	Reported	Substantiated	Of which organic wastes
England and Wales	32409	20158	2129
Scotland	. .	2878	. .
Northern Ireland	2881	2055	512
Total	35290	25091	2641

Source: Environment Agency 1998 (1996 figures).

The total figure for organic wastes, 2641, is an underestimate in the sense that figures for Scotland were not available. However, the inaccuracy is lessened in that most, but not all, organic waste incidents are caused by agriculture.

4.8 Soil resources

The agricultural sector is particularly vulnerable to soil erosion, because arable production removes the protective vegetative cover from the soil. There are on-site and off-site costs to soil erosion. On-site costs affect crop yields: both wind and water sweep away seeds and the topsoil that contains nutrients. The soil washed into water courses is often the main source of both the nutrient phosphorus and pesticides which are bound to the soil particles. Thus off-site costs include the increased costs of water treatment when soil and eutrophication clog up water filters, discolour water, and introduce toxic chemicals. This means that the calculations for the soil category (and not the water category) will incorporate the increased costs of water treatment caused by nutrients attached to soil particles. Soil erosion also increases flood damages.

Volker (1998) uses data from monitoring programmes by the Soil Survey of England and Wales and estimates the on-site costs of soil erosion by examining water erosion on wheat production. The estimates are made using wheat because of the lack of alternative data.

Table 8: Rates of soil erosion in England and Wales

Tonnes per ha per year

Erosion rates	Soil loss
Low	3.1
Moderate	7.2
High	11.5

Source: see text.

The low erosion rates are used in conjunction with the estimated percentage reduction in yields. For deep soil it is estimated that there is a 0.4% reduction in yield per mm of soil eroded. For shallow soil the figure is a 2% reduction in yield per mm of soil eroded. This information is combined with the area of agricultural land in England and Wales that is at moderate to high risk of erosion, in order to obtain a conservative estimate. That area totals 2,404,200 ha. Unfortunately there are no matching data for Scotland and Northern Ireland – however only 11% of agricultural land in Scotland and 7% of agricultural land in Northern Ireland is cropland, so the margin of error does not prevent the use of this data in this study.

The following sections contain estimates of the physical environmental services from natural capital.

4.9 Land use, agri-environmental and protected area schemes

Table 9 Land use in the UK

(Thousand ha)

Country	Agricultural land	Forest and woodland
England	0.73% of 12972 = 9469.56	0.08% of 12972 = 1037.76
Wales	0.79% of 2064 = 1630.56	0.12% of 2064 = 247.68
Scotland	0.75% of 7710 = 5782.50	0.15% of 7710 = 1156.50
Northern Ireland	0.81% of 1348 = 1091.88	0.06% of 1348 = 80.88
Total	0.74% of 24093 = 17828.8	0.10% of 24093 = 2409.3

Source: Digest of Environmental Statistics 1998 (1996 figures). Note: calculated using percentage of country area.

Agri-environmental schemes are voluntary, and consist of contracts between farmers and the government. These contracts purchase changes in production practices and thereby induce private provision of public goods, and reduce negative externalities (Bonnieux et al 1998). In contrast, the Forestry Commission as the caretaker of public forest land, is required to provide public goods, the provision of which is funded by its commercial plantations. The estate includes approximately 175,000 ha within National Parks, National Scenic Areas and Areas of Outstanding Natural Beauty. There are not many schemes into which private forestry can enter, only perhaps ESAs or SSSIs. However, the take up is not known. This highlights a problem with the data below. The figures for ESAs and SSSIs may, or may not be overestimates, because information is not available as to the percentage of land area for both schemes that is not on agricultural or forestry land. A second problem is that some land may be entered into more than one scheme.

Table 10 Agri-environmental and protected area schemes in the UK

(Thousand ha)

Scheme	Area of land	% of UK land area
Environmentally Sensitive Areas	3377	14.02
Countryside Stewardship Scheme	127.75	0.53
Organic Aid Scheme	139.24	0.58
Habitat Scheme	14.72	0.06
Farm Woodland Premium Scheme	51.00	0.21
Nitrate Sensitive Areas	28.00	0.12
Sites of Special Scientific Interest	2084	8.65
Forestry Commission	175	0.73
Total	5996.71	24.89

Sources: Agriculture in the UK 1998 (MAFF) (1998 figures), except SSSIs and ESAs which are taken from Digest of Environmental Statistics 1998 (1997 figures), and Forestry Commission protected areas, which are taken from its Annual Accounts 1997-98. Agri-environmental schemes are part-funded by the EC.

There are also land-use changes. The area of the UK covered by forest has been increasing steadily over the last few decades. From 1980 to 1990, the rate of increase was 1.3% per year. From 1990 onwards the rate of increase slowed to 0.63% per year. Data for new forest plantings only extends up until 1996, but using the calculated rate of increase for the 1990s, the data can be estimated. The estimated increase in forest between 1998 and 1999 is therefore (2551.3–2535.4) which is 15.867, or 15867 ha.

Table 11: Forest cover in the UK

(Thousand ha)

1990	1991	1992	1993	1994	1995	1996	1997	1998	1999
2412	2427	2439	2456	2470	2485	2504	2519.7	2535.4	2551.3

Source: Figures up to 1996 taken from Forestry Commission, figures 1997-9 calculated using rate of increase.

The land that is sold from agriculture or forestry into urban use is not recorded in these accounts as a loss since any change in its state should be recorded as such in the urban sector's accounts.

4.10 Hedgerow loss

The removal of hedgerows in England and Wales is undertaken by the agricultural and forestry sector to enlarge areas of land in production. There are no data for Scotland and Northern Ireland, which means that the final figure will be a small under-estimate.

Table 12 Length of hedgerows in England and Wales

(Thousand km)

1990	1993	1994	1995	1996	1997	1998	1999
431.8	377.5	361.0	345.1	330.0	315.6	301.7	288.5

Source: figures up to 1993 taken from Digest of Environmental Statistics 1998, figures 1994-9 calculated using rate of loss.

The rate of loss between 1984 and 1990 was 4.3% per year, meaning that the length of hedgerows in England and Wales decreased from 563,100 km to 431,800 km. From 1990 to 1993 the rate of loss remained almost identical at 4.4%. This figure is used to estimate the length of km lost in 1998. As can be seen from table 12, the length of hedgerows lost between 1998 and 1999 is (301.7-288.5) which is equal to 13.2, or 13,200 km. The data does not include privately grown hedges in England and Wales, and most remaining hedgerows are on agricultural or forestry land. Hence the figure is a fairly accurate estimate of the loss to England and Wales to be entered into the sectoral account.

4.11 Biodiversity Resources

Data detailing the damage to biodiversity specifically from agriculture and forestry are difficult to find. There are multiple explanations of species' population decline, and these explanations often interact. If one species declines, it has an effect on all other species in the food chain. If agriculture or forestry then affects one of these other species in the food chain, it is problematic attributing damage to the sector relative to another sector. Data that is available is damage caused to Sites of Special Scientific Interest. Even then, damage is not entirely attributable to agriculture or forestry. The damage is measured in ha, which demonstrates the point that landscape and biodiversity resources are inextricably linked via habitats.

Table 13: Annual physical damage to SSSIs

(Ha)

Type of physical damage	Area damaged
Loss, total or partial	700
Long term damage (40 years)	4,800
Short term damage (10 years)	26,500
Total	32,000

Source: Adger and Whitby (1996), 1991 figures.

5 The shadow prices

The standard methods of estimating willingness to pay, such as hedonic pricing, travel-cost method, and contingent valuation, apply to the evaluation of environmental services. These methodologies are not discussed here - see, for example, Garrod and Willis (1999) for detailed descriptions.

5.1 Air resources: global, regional and local

Estimating the marginal damage cost of atmospheric gases is problematic because the flow of emissions that enter the atmosphere add to a stock that dissipates at varying rates. To estimate the damage costs for CO_2, CH_4 and N_2O requires modelling the relationships between emissions, radiative forcing and damage. The models estimate the present value of damage caused in the future, when a doubling of pre-industrial atmospheric concentrations has occurred, by a unit of emissions released now. This means that a discount rate must be chosen – and it is this choice which largely, though by no means entirely, accounts for the difference in the estimates in the literature.

SO_2 and NO_X are regional pollutants because they are carried downwind of the polluting source. Thus, the relevant marginal damage costs must be the sum of the marginal damage costs for all the countries affected. This applies not only to evaluating the damage caused by emissions deposited in other countries, but also to evaluating the damage caused locally. Table 14 summarises the marginal damage costs used in this study.

Table 14: Global, regional and local marginal damage estimates for air pollutants

(£ per tonne)

Pollutant	Impact	1998 prices
CO_2 (as C)	Global warming	**29.8**
CH_4	Global warming	**77.9**
N_2O	Global warming	**2961.2**
SO_2	Direct acute mortality	3879.6
	Direct morbidity	3.1
	Indirect acute mortality (via sulphates)	1272.7
	Indirect morbidity (via sulphates)	584.0
	Crop damage	8.2
	Materials damage	329.5
	Ecosystem damage	12.3
	Total SO_X	**6089.3**
NO_X	Indirect human health (via ozone)	- 603.5
	Indirect acute mortality (via nitrates)	1271.6
	Indirect morbidity (via nitrates)	622.0
	Direct crops	- 121.1
	Crops (via ozone)	- 185.8
	Total NO_X	**983.2**
PM_{10}		
	Acute mortality	2680.8
	Morbidity	1193.6
	Total PM_{10}	**3874.4**
NMVOCs		
	Cancer (via benzene)	139.6
	Human health (via ozone)	727.7
	Crops (via ozone)	420.8
	Total NMVOCs	**1288.1**

Notes: CO_2, CH_4, N_2O marginal damage costs taken from Eyre et al (1997). They are converted from US dollars to pounds using September 1999 exchange rates. SO_2, NO_X, PM_{10} and NMVOC damage estimates taken from Pearce et al (1998). All prices are adjusted using GDP deflator taken from UK National Accounts 1998. The discount rate used here is 2 per cent in real terms and is applied to the greenhouse gas damage estimates. The 2% discount rate does not include the private rate of time preference, it solely reflects societal values. .

Marginal damage to buildings from sulphates is calculated by constructing various scenarios of damage costs that occur with different quantities of acid deposited. Acute mortality and morbidity are estimated from epidemiological dose-response functions, and the value of a statistical life. The marginal damage cost for PM_{10} is evaluated using the same methodology. For crops, nitrate deposition acts as a fertiliser, and over all, marginal benefits are recorded from field studies, even when costs from ozone and sulphate damage are deducted. Note that crop damage is included in the total shadow price, rather than assumed to be reflected in the GDP, because farmers are assumed to be producing on the production possibilities frontier. Some health effects from NO_X, and crop damage from NO_X and NMVOCs occur via ozone.

5.2 Water resources

Green and Tunstall (1991) estimated households' WTP to improve the quality of freshwater from ‘medium’ to ‘good’, and from ‘poor’ to ‘medium’. These WTP estimates are used as marginal damage estimates for decreasing biological water quality. The WTP is multiplied by the relevant kilometre distances, and then multiplied by 2.49 million. This is the number of rural households in the UK. As estimates in this paper are confined to use values, only the households in proximity to rivers and canals are counted.

Marginal damage estimates for eutrophication caused by the riverine discharge of nutrients into coastal areas present problems. The only coastal and marine study estimating eutrophication costs appears to be that of Turner et al (2000). Contingent valuation and travel-cost studies estimated the WTP to reduce by 50% the discharge of nutrients into the Baltic Sea. Dividing total WTP by 50% of the figure for total tonnes discharged of N and P, and allocating damage according to the ratio of N to P discharged, gives marginal damage estimates of around £8000 per tonne. As shown later, this produces an extremely high damage estimate of nearly £2 billion. We regard this figure as unsatisfactory and eutrophication damage requires further research.

There are few studies estimating the marginal damage from pesticides released into the environment. Two factors help to explain this absence. Firstly, little is known about the individual toxicity levels of pesticides. Secondly, little is known about how pesticides interact with each other, and with other chemicals. It is also difficult to match physical data with the data on marginal damages. Thus the more plentiful physical data measuring concentration levels cannot be used. Instead data measured in tonnes must be used, because the only European study estimating WTP for pesticide reductions uses units of kg. Foster et al (1998) estimate the WTP to avoid pesticide residues in food using the contingent ranking method. The WTP incorporates not only concern for human health, but also concern for the safety of birds, the latter of which turns out to be large relative to health risks. Their valuation of £12 per kg of pesticide is crude, not least because pesticides vary substantially in their toxicity. The 25,000 tonnes of pesticides released annually into the UK environment would, at £12 per kg, produce a potential externality of £300 million. However, this is at best a benchmark, since, apart from varying toxicity and having just one study on willingness to pay to avoid pesticide risks, the amount of pesticide actually reaching the environment is some unknown fraction of 25,000 tonnes. For these reasons we confine the damage estimate to water-related pesticides, but acknowledge that this is likely to produce a serious underestimate of true damage.

Water pollution incidents occurring within the agricultural sector have been targeted by subsidies in recent years in an effort to reduce damage caused. Between 1989 and 1994, £300 million was spent on installing farm waste facilities. Pearce (1999) uses this figure to give a lower estimate for the benefits from avoided incidents. The £300 million spent was followed by a 10% reduction in the number of organic waste incidents, and it is assumed that these avoided incidents stretch into the future. Using a discount rate of 5%, £300 million saves a total of approximately 2500 avoided incidents. The cost of an avoided incident is therefore £120,000. However this figure is taken from control costs, and although can be interpreted as a lower estimate of the general public’s maximum WTP to reduce incidents, it is not a direct measure of damage. We later consider the estimates derived in Pretty et al. (2000) for water pollution incidents.

Table 15 Marginal damage estimates for water pollution

Indicator	Marginal damage estimate	
		1998
Biological water quality	Medium to good quality, per hh per km	£0.0016
	Poor to medium quality, per hh per km	£0.0029
Discharges of nutrients	per tonne N	£7594.9
	per tonne P	£8210.7
Discharges of pesticides	per kg pesticide	£12.3
		1998
Water pollution incidents	per incident	£134,276.4

Note: hh denotes household. Figures in text updated to 1998.

Many studies evaluate the costs of nitrates in drinking water by using Hanley's (1991) WTP estimate to reduce nitrate concentrations below 50 mg per litre. However, recent data for drinking water (Digest of Environmental Statistics, 1998 (1996 figures) indicates that elevation above 50 mg per litre is infrequent.

5.3 Soil resources

Volker (1998) sets up a model that calculates the present value of profits from farming, comparing the situation with erosion to the situation without. Output is reduced by water erosion. Soil is a form of natural capital, providing a flow of services over time. Therefore, once soil is depleted, the productive capacity lost decreases profits into the future. Three different discount rates are used in turn to evaluate water erosion: 1%, 3%, and 5%, both for the losses on deep and shallow soils. The final figure used in this study's accounts will have been calculated using a 3% discount rate in order to be as consistent as possible with the greenhouse gas damage estimates. Evans (1996) calculates the aggregate wind erosion and off-site costs. A conservative estimate of costs to the water industry shall be taken. Unfortunately, the wind erosion costs do not derive from a present value model and so are likely to be an underestimate.

Table 16 The on-site and off-site costs of erosion

	Marginal damage costs: 1998 prices, 3% discount rate	Total costs: 1998 prices £ million
Water erosion:		
Deep soils	£1.799 present value per ha	£ 2.16
Shallow soils	£8.996 present value per ha	£ 10.81

Total	£5.398 present value per ha	£ 12.97
Wind erosion		£ 0.84
Property and road damage		£ 2.45
Costs to water industry		£ 4.22-31.68

Notes: The total for water erosion is calculated making the simplifying assumption that of the 2,404,200 ha of land at risk from soil erosion, 50% is deep soil, and 50% is shallow soil.

5.4 Landscape and biodiversity resources

Bullock and Kay (1997) conduct a contingent valuation survey asking for the WTP of rural households for policies that preserve extensified agricultural landscapes. There are problems of double counting when using these estimates. When a respondent gives a WTP estimate for the agricultural landscape, they are likely to visualise not only arable fields, but also hedgerows, small farm woodlands and the natural wildlife habitats contained within. It would be difficult to separate the elements of that perception. The same arguments apply to the evaluation of land converted to agri-environment schemes. Therefore evaluating changes in natural capital for this category will mean simultaneously evaluating those for the biodiversity category. But going back to agricultural landscapes, the WTP used may be an overestimate if it is perceived that agricultural land has lost value through intensification. The WTP estimate is multiplied by 2.49 million households, the estimated number of rural households in the UK.

WTP estimates are not available for each type of agri-environmental and protected area scheme, only for ESAs and SSSIs. The WTP for SSSIs is in units of hectares, therefore it will also be applied to evaluating the remaining schemes, since these have similar environmental objectives. However, this also poses a risk of double counting since many schemes might be considered substitutes for each other. If land devoted to one scheme was converted to urban use, for example, part of its value to visitors could simply be transferred to other land (Adger and Whitby 1991). The WTP for ESAs is multiplied by 8.26 million, the estimated number of households visiting ESAs every year (calculated from Garrod and Willis (1995).

Much forest and woodland is not legally accessible since it consists of commercial plantations. It is not known what percentage of forest and woodland is accessible to the general public. However, Bateman and Langford (1995) estimate the WTP for recreational woodland walks per person, and not hectares. Thus the problem of physical information is no longer relevant, for respondents give WTP according to their surrounding woodland. However, this does mean though that the additions to the stock of forest and woodland will only be valued indirectly through changing WTP values. It will depend on whether greater opportunities for forest recreation increase demand by more than the increase in supply, or whether demand becomes saturated. The WTP value is multiplied by 6.17 million, the estimated rural population of the UK.

Table 17 WTP estimates for landscape and biodiversity

Land use	Subject of WTP study	WTP estimate

Agricultural landscape	Policies for extensified, agricultural landscape	1997 prices £55 per hh per year	1998 prices £56.4
Forest and woodland	Woodland walks	1995 prices £12.55 per person per year	1998 prices £13.7
Environmentally Sensitive Areas	WTP additional taxes for South Downs ESA	1992 prices £19.47 per visitor hh per year	1998 prices £22.7
Sites of Special Scientific Interest	Wildlife in an SSSI in Upper Teesdale	1986 prices £41.6 per ha per year	1998 prices £69.5

Sources of estimates: Garrod and Willis (1995), Willis and Benson (1988).

Finally, these accounts deduct from the above valuation of landscape and biodiversity resources the damage to those same resources. Here, damage to landscape and biodiversity is solely valued via damage to SSSIs since other data are not possible to evaluate. Adger and Whitby (1996) follow English Nature's advice on the duration of short term damage (ten years) and long term damage (forty years). Using the same WTP estimate as that for evaluating environmental services from SSSIs above, the flow of lost benefits is calculated, discounted at 3%. The total figure is not consistent with the other calculations in this study because of the method by which the authors adjust the 1988 WTP estimate to 1991 prices. Added to the rate of inflation is an assumed income elasticity of demand for environmental goods. We have not made this adjustment elsewhere than here. The implication is that the damage to SSSIs will be an overestimate relative to the figures in the rest of the accounts.

Table 18 Annual monetary damage to SSSIs

(£ million)

Value of damage	1991 prices	1998 prices
Loss	1.19	1.44
Long term damage	4.71	5.72
Short term damage	10.59	12.85
Total Damage (discounted at 3%)	31.34	38.03

As explicitly shown for the pesticide shadow price, WTP estimates for other resources are likely to include an element of damage to land and biodiversity resources.

6 The environmentally adjusted accounts

Table 19 presents our revised set of accounts for UK agriculture.

Table 19 Adjusted Net Domestic Product for UK Agriculture and Forestry 1998

£1998 prices, million

Gross Output at basic prices	**16,870.7**
of which:	
Agriculture	16,414.0
Forestry	456.7
Gross Value Added at basic prices	**7,651.1**
of which:	
Agriculture	7,336.0
Forestry	315.1
Net Value Added at basic prices	**5,054.5**
of which:	
Agriculture	4,787.0
Forestry	267.5
Net Value Added at factor cost	**5,292.5**
of which:	
Agriculture	5,025.0
Forestry	267.5
Net Value Added at market prices	**2,619.5**
of which:	
Agriculture	2,352.0
Forestry	267.5
Environmental Services	**594.9**
of which:	
Agricultural landscape	140.7
Forest and Woodland	84.5
Environmentally Sensitive Areas	187.6
Sites of Special Scientific Interest	182.1
Depreciation of Natural Capital	**1,072.2**
of which:	
Air resources:	**585.4**
of which:	
Global:	**318.2**
Net CO_2 emissions	-71.5
CH_4 emissions	98.6
N_2O emissions	291.1
Regional:	**13.2**
SO_2 emissions	5.4
NO_X emissions	7.9
Local:	**254.0**
SO_2 emissions	9.1
NO_X emissions	4.4
PM_{10} emissions	11.2
VOC emissions	229.3
Water resources:	**428.3**
of which:	

Biological water quality	70.5
Discharges of nutrients	na. (see text)
Discharges of pesticides	3.2
Water pollution incidents	354.6
Soil resources:	**20.5**
of which:	
Water erosion	13.0
Wind erosion	0.8
Off-site costs	6.7
Landscape and biodiversity resources:	**38.0**
of which:	
Damage to SSSIs	38.0
Adjusted Net Domestic Product	**4,577.2**

Notes: Figures for Forestry gross output and gross value added are for the year 1997, communicated via telephone by the ONS. The figures are adjusted to 1998 prices using the relevant GDP deflator. The figures for fixed capital consumption, in order to reach net value added, are only available for public sector forestry. These are taken from the Forestry Commission's 1998 accounts. Man-made capital depreciation is 15.1% of gross value added. Therefore, gross value added for the entire forestry sector is multiplied by 0.849 to reach net value added. However, this method of estimation assumes private forestry is under identical management This is not the case, and so the percentage error may be greater than 100%. Omitting private forestry depreciation altogether however would also entail a large error, because private forestry contributes the larger proportion of forestry's gross value added.

Net Value Added at market prices is a proxy for conventional NDP at 'world prices'. The proxy is reached by deducting production linked subsidies of £2.4bn (1998 prices) from Net Value Added at basic prices. So conventional NDP at world prices is equal to £2.6 billion. The more accurate method of re-valuing output using the set of p'_{WORLD} prices is not possible since data are not detailed enough. However, we go back to Net Value Added at basic prices to add environmental services and subtract depreciation, since these values would change significantly if subsidies of £2.4 billion were taken away from the agricultural sector. And so adjusted NDP is equal to £4.6 billion.

We have chosen not to record a value for eutrophication. Use of the implied figures in the Turner et al (2000) study for the Baltic produces eutrophication damage of some £1.94 billion which, while not impossible, seems to us to be an exceedingly high number. Nor is there any real rationale for 'transferring' this figure from the Baltic context to the UK.

Net carbon emissions were negative. However the negative net figure could have been larger (though likewise possibly smaller), for the accuracy of soil measurements can vary by $\pm 50\%$. In addition, the extent of the monetary damage from greenhouse gases varies considerably depending on the discount rate chosen. The discount rates chosen were all around 2%, a figure derived mainly from the growth rate of real consumption per capita multiplied by an estimate of the elasticity of the marginal utility of income function (Pearce and Ulph, 1999).

Damage to rivers and canals, valued via biological water quality is also a potential overestimate. The agricultural and forestry sector's fertiliser use cannot be held entirely responsible for low biological water quality. The OECD estimates of the proportion of nutrients emanating from the sector show that its fertiliser use can not be entirely to blame. However, it is possible, if the decrease in biological water quality from slurry and silage incidents is included, that the sector is nearly entirely responsible. This introduces the possibility of double counting into the accounts. A second problem is the assumption that pollution in rivers and canals is a flow that does not contribute to a stock in these rivers and canals. Refining this assumption by modelling the change in water quality from year to year would reduce the damage estimate.

On the other hand, other elements of depreciation were underestimated. Soil erosion in Scotland and Northern Ireland was not evaluated. Conservative values were chosen for the area at risk of erosion, the rate of erosion, and costs to the water industry. The wind erosion value is also an underestimate since its calculation did not include future productivity losses from the soil eroded.

Finally we note again that the non-use and option values conferred on the countryside are not estimated due to the uncertainty inherent in deriving such figures in the absence of readily applicable data. While there are several studies of willingness to pay to conserve natural habitat in the UK, none appears to relate to agriculturally managed countryside. But there is no doubt that as far as the benefits of the agricultural sector are concerned, one relevant value is the willingness to pay of the population at large to conserve the countryside without actually making use of it or intending to use it, plus the option value of city-dwellers desiring the opportunity to live in the countryside in the future. It seems likely that these values could be substantial. Thus, Dillman and Bergstrom (1991) conducted a contingent valuation study for limited areas of prime agricultural land in Greenville County, South Carolina and found individual valuations of $5.7 to $8.9 per household for a range of 18,000 to 72,000 acres. These valuations reflect household willingness to pay to retain the relevant land as agricultural land rather than allowing it to be converted to residential development. The WTP function was a simple linear form, WTP = $4.6 + 0.06 ACRES where acres are measured in 1000 acre units. Clearly, there would be no justification in transferring such a value function to the UK and we would conjecture that, in any event, there would be a strong distance-decay relationship whereby distant countryside is valued less than countryside near to towns.

Willis and Whitby (1985) conducted a contingent valuation study of willingness to pay to conserve green belt land round urban areas. They derive a mean willingness to pay of some £327 per hectare, or, aggregated across green belt land in 1988 (1.55 million ha), just over £500 million, or some £650 million in 1998 prices. But the problem is that green belt land is almost certainly valued for its 'natural' qualities more highly than other agricultural land due to its proximity to urban conurbations. Using these figures to derive illustrative non-use values would also run into potential double-counting problems when combining them with the environmental service values already in the accounts. As an experiment, we multiply the valuation per household for policies that preserve extensified agricultural landscapes, used in the section 'landscape and biodiversity resources' by non-rural households. The resultant 'non-use' value is £1.2 billion.

Our conclusion, conditional on further work, is that UK agriculture has a 'true' or 'green' Net Domestic Product of about £4.6 billion, a modest decrease on the estimate of net value added at basic prices (£5.1 billion),. But were our illustrative figure for non-use value to be included, there would be a modest *increase* on the estimate of net value added at basic prices to £5.8 billion. The main adjustments are shown in Table 20 for convenience.

Table 20 Summary of Adjusted Agricultural Net Domestic Product

(£ million, 1998)

Net Value Added at Basic Prices	5,054.5
+ Environmental Services	+ 594.9
- Environmental depreciation	- 1,072.2
= Adjusted Net Domestic Product	**4,577.2**

7 Comparisons with other studies

Two other studies have been carried out for the UK. The first, by Adger and Whitby (1993) attempts a similar exercise to the one here but for the 'land use sector' as a whole, i.e. included protected areas that may or may not be associated with agricultural activity. The second, by Pretty et al. (2000), is confined to estimating externalities without incorporating them into a set of revised accounts.

The results of the Adger-Whitby study are shown in Table 21. We have adjusted the estimates which were for 1988 but at 1993 prices, to 1998 prices.

Table 21 Adger-Whitby modified UK land use accounts

(£ million)

	1993 prices	1998 prices
Net Domestic Product	**4,028.0**	**4,573.0**
Plus		
Environmental services	**888.0**	**1,008.1**
plus		
Appreciation of natural capital	**121.4**	**137.8**
of which:		
Net CO_2 emissions	+ 132.2	+ 150.1
Nitrates in drinking water	- 10.8	- 12.3
minus		
Defensive expenditures	**58.0**	**65.8**
(= maintenance of SSSIs)		
Adjusted Net Domestic Product	**4,979.4**	**5,653.1**

Note: we have revised the plus/minus notation in Adger-Whitby so that + means appreciation and - is depreciation.

The Adger-Whitby calculations result in an adjusted NDP that is larger than conventional NDP, due to the net appreciation of natural capital outweighing degradation. This is not in keeping with the result in our own study, unless we include our illustrative non-use value, but we also believe we have understated overall depreciation of natural capital. The Adger-Whitby study does make some attempt to include option and non-use values for 'greenbelt' areas round towns of £642 million in 1988. However, their other positive values for environmental capital come from protected area or recreationally designated areas which means that they are not strictly recording the values residing in the agricultural sector alone. Indeed, this is why Adger and Whitby refer to it as the 'land-use' sector. We have been unable to find values for non-use and option values for 'the countryside' in general but suspect that it is highly positive. We regard our £1.2 billion as notional and almost certainly an underestimate for agricultural land, let alone for the countryside in general.

The Adger-Whitby figures for depreciation of natural capital are made up of an appreciation of capital due to a net carbon balance, and a small negative adjustment for nitrate pollution. The figures for net CO_2 do not incorporate all the soil fluxes. Other forms of depreciation are not included beyond those recorded as defensive expenditures, i.e. the cost of maintaining SSSIs against damage much of which is due to agricultural practice. As it happens, there is a theoretical dispute concerning the validity of deducting defensive expenditures. Adger and Whitby side with those who believe the expenditures should be deducted, but theoretical models of income accounting do firmly suggest that defensive expenditures should be ignored as intermediate expenditures (Mäler, 1991).

Table 22 shows the results obtained by Pretty et al. (2000).

Table 22 Pretty et al. estimates of annual external costs of UK agriculture, 1996

(£ million, range values for 1990-1996)

Cost Category	**UK (£ million)**	**Range[1]**
1. Damage to Natural Capital: Water		
1a Pesticides in sources of drinking water	120	84- 129
1b Nitrate in sources of drinking water	16	8- 33
1c Phosphate and soil in sources of drinking water	55	22- 90
1d Zoonoses (esp. *Cryptosporidium*) in sources of drinking water	23	15- 30
1e Eutrophication and pollution incidents (fertilizers, animal wastes, sheep dips)	6	4- 7
1f Monitoring and advice on pesticides and nutrients	11	8- 11
2. Damage to Natural Capital: Air		
2a Emissions of methane	280	248- 376
2b Emissions of ammonia	48	23- 72
2c Emissions of nitrous oxide	738	418-1700
2d Emissions of carbon dioxide	47	35- 85
3. Damage to Natural Capital: Soil		
3a Off-site damage caused by erosion[2]	14	8- 30
3b Organic matter and carbon dioxide losses from soils	82	59- 140
4. Damage to Natural Capital: Biodiversity and Landscape		
4a Biodiversity/wildlife losses (habitats and species)	25	10-35
4b Hedgerows and drystone walls	99	73-122
4c Bee colony losses	2	1-2
4d Agricultural biodiversity	+	+

5. Damage to Human Health: Pesticides 5a Acute effects 5b Chronic effects	 1 +	 0.4-1.6 +
6. Damage to Human Health: Nitrate	0	0
7. Damage to Human Health: Micro-organisms and Other Disease Agents 7a Bacterial and viral outbreaks in food 7b Antibiotic resistance 7c BSE and new variant CJD	 169 + 607	 100-243 + 33-800
TOTAL	**£2343**	**£1149-3907**

Notes: 1. The ranges for costs do not represent formal standard deviations of the data as this is impossible given the huge variation in types of data and contexts. The ranges represent best estimates for higher and lower quartiles for costs incurred annually during the 1990s. The range values for the external costs in Category 2 are calculated from the ranges stated in studies of external costs of each of these gases, rather than the variation of emissions during the 1990s. 2. The offsite damage caused by erosion in category 3a does not include the costs of removing soils/sediments from drinking water (these are in cost category 1c). 3. BSE costs are an average for 1996 and 1997. 4. This table does not include private costs borne by farmers themselves. 5. + means not yet able to calculate costs.

The Pretty et al (2000) estimates are interesting. Once again, comparison with our own estimates is difficult because of differences in the methodological approach. First, the Pretty et al. estimates make no attempt to include positive non-marketed benefits, so that the closest comparison that can be made is between our negative externality estimate of £1.07 billion and theirs of £2.3 billion (we ignore the difference in year estimates, the Pretty et al. estimates being for 1996 and ours for 1998). Secondly, Pretty et al, include a substantial element for food risks and BSE. It is unclear if these constitute externalities that should be debited to the agricultural sector, but in so far as the problems arise from mismanagement of agriculture it seems legitimate to include them as social costs. They do not, however, appear to be environmental externalities, the focus of our own analysis. A fairer comparison with our own estimates, therefore, would be to take the Pretty et al. estimates for their items 1-6, which comes to £1.6 billion. Third, the Pretty et al estimates are derived in a different fashion to ours. We have sought to find willingness to pay estimates of damage, whereas Pretty et al. have mainly sought to find costs imposed on others in the form of restoration or reinstatement costs to return the environment to the original state. Such costs are not damage costs in the economic sense unless there is some absolutely binding constraint to ensure such restoration. In terms of the sustainability literature, their approach is based on 'strong sustainability' -i.e. whatever it costs to reinstate the environment to its pre-damage condition - whereas ours is based on a 'weak sustainability' approach which is consistent with cost-benefit analysis.

One major discrepancy is in the estimate of damage from the greenhouse gas nitrous oxide. The Pretty et al. estimates suggest a unit value of £7530 per tonne N_2O, whereas we use £2961 per tonne. The source for these estimates is actually the same - Eyre et al (1997). Inspection of the Pretty et al. workings shows that they have taken the entire range of estimates in Eyre et al (1997), rather than selecting a single representative model and a consistent discount rate. Their approach is to take the lower end of the range and then add one quarter of the difference between the low and high estimates in the range, i.e. L + 0.25 (H-L) where L is the low estimate and H is the high estimate. This gives a unit value 2.5 times that used here. We prefer our own approach since the lower and upper end of the ranges in Eyre et al. (1997) are due mainly to differing discount rates. Arguably, the procedure adopted by

Pretty et al. emerges from a crude adjustment for an 'expected value' of a discount rate, but we have preferred to be consistent as possible in using a 2 per cent discount rate. In the same way, the Pretty et al. (2000) estimates for CO_2 and CH_4 are factors of three and two above ours because of the same methodological difference.

An omission in our accounts is ammonia and hence the Pretty et al. estimate could be legitimately added to our accounts, say some £50 million. We note also that we have secured a net carbon gain in our study whereas Pretty et al. have a net carbon loss.

Overall, the Pretty et al. estimates have the most comprehensive coverage whilst ours, we would argue, are nested in a theoretically more sound methodology. Despite these differences, the two studies are fairly consistent in suggesting that agricultural externalities are of the order of £1-2 billion per annum in the UK. Given the modest size of the sector, some £5.1 billion in net value added terms gross of production-linked subsidies, and perhaps only £2.6 billion when 'revalued' at world prices, these externalities amount to a massive offset against the contribution that agriculture makes to the UK economy. Offsetting this, we find at least £0.6 billion environmental services from the sector in terms of use values and a tentative £1.8 billion in all once non-use and option values are included. If so, the result is highly supportive of those who argue that modern agricultural sectors have a major part of their justification in the provision of environmental services rather than food.

Annex 1 National accounting concepts

Definition of the agricultural and forestry sector:

The agricultural industry is defined as the collection of local units that carry out agricultural activities including both farms and specialist agricultural contractors. It includes any inseparable non-agricultural activities that these units carry out. It excludes services relating to design planting and maintenance of gardens, parks, and green areas for sports facilities. It also excludes units producing solely for their own consumption.

The forestry industry comprises both privately and publicly owned plantations. The Forestry Commission has overall responsibility for the publicly owned plantations, which cover approximately 35% of total forest and woodland in the UK. Management of these plantations has been delegated to the agency Forest Enterprise. The main product from both types of plantation is timber, but recreation is also marketed in the form of camping sites, environmental schemes and hunting.

Definition of the terms:

Gross Domestic Product (GDP): measures the output produced by factors of production located in the domestic economy regardless of who owns these factors. GDP measures the total value added within an economy.

Gross National Product (GNP): measures total income earned by domestic citizens regardless of the country in which their factor services were supplied. GNP equals GDP plus net property income from abroad. GDP is the basic concept used in this paper.

Net Domestic Product: NDP is calculated by subtracting depreciation of man-made capital from GDP.

'Basic prices': amount received by the producer for a unit of goods or services, minus any taxes payable, plus any subsidy receivable on that unit as a consequence of production or sale. Thus NDP at basic prices includes subsidy receipts.

Market prices: amount paid by the purchaser, which includes any indirect taxes such as non-deductible VAT, and does not include indirect subsidies.

Thus the difference between GDP (NDP) at basic prices and GDP (NDP) at market prices in agriculture is the aggregate sum of subsidies linked to sales. There are no subsidy programmes for forestry. Therefore there is no difference between GDP at basic prices and GDP at market prices for forestry.

Factor cost: GDP at market prices exceeds GDP at factor cost by the amount of revenue raised in indirect taxes, net of any subsidies on goods and services.

Indirect taxes are negligible for agriculture and forestry. Hence, for agriculture, GDP at market prices is actually exceeded by GDP at factor cost by the aggregate sum of subsidies linked and non-linked to production. Therefore, the difference between GDP at basic prices and GDP at factor cost is the aggregate sum of subsidies non-linked to sales. This latter type of subsidy includes animal disease compensation, and land set-aside. There is no difference between GDP at market prices and GDP at factor cost for forestry.

The relevant concept for this study is GDP at market prices. This is the same as valuing GDP using OECD producer subsidy equivalents (PSEs). It is an approximation to valuing output at p'_{WORLD} (see Figure 1), in the absence of detailed and accurate data on current world prices for agricultural commodities. The most obvious inaccuracy is that deducting both forms of subsidies to reach GDP at market prices, has still not deducted the difference between p_S and p'_{WORLD} multiplied by UK domestic demand. This is also a form of subsidy from the UK consumer to the UK agricultural sector.

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